Life Cycle Analysis of Greenhouse Gas Emissions from the Mining and Milling of Uranium in Saskatchewan

A thesis submitted to the College of
Graduate Studies and Research
in partial fulfillment of the requirements
for the degree of Master of Science
in the Department of Civil and Geological Engineering
University of Saskatchewan, Saskatoon

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Abstract

This thesis presents a detailed study of life cycle greenhouse gas (GHG) emissions intensity during the uranium mining-milling phase of the nuclear fuel cycle for three paired uranium mine-mill operations in northern Saskatchewan (SK). The study period runs from 2006 – 2013 for two of the three pairs, and from 1995-2010 for the third. The life cycle analysis has been conducted based on the ISO 14040:2006 standard using a Process Chain Analysis methodology.

This study differs from previous studies of GHG emissions intensity during the uranium mining-milling phase of the nuclear fuel cycle in two key respects. First, it has a very large system boundary which includes the uranium exploration and mine-mill decommissioning phases. Second, it utilizes a life cycle inventory database to include many processes which would normally fall outside of the system boundary due to their small individual contributions. These differences contribute to a more accurate result.

The production-weighted average life cycle GHG emissions intensity is estimated as 45 kg CO_2e/kg U_3O_8 at an average ore grade of 9.12% U_3O_8 based on relative U_3O_8 production volumes at Mine-Mill A, B, and C from 2006 to 2010. The 95% confidence interval for the production-weighted average result ranges from 42 to 49 kg CO_2e/kg U_3O_8, indicating that overall uncertainty in the result is low. Life cycle GHG emission intensity for the three uranium mine-mill pairs are 84, 66, and 35 kg CO_2e/kg U_3O_8 at average ore grades of 0.71%, 1.54%, and 11.5% U_3O_8 respectively.

Nearly 90% of life cycle GHG emissions are associated with operation of the uranium mine-mills, primarily from energy consumption during operation (69% of total) transport of materials and personnel (7.0%), and use of reagents (5.6%). Remaining processes each individually account for less than 5% of the total.

In calculating emissions from electricity consumption, the base-case emission intensities reported above use a province-wide electricity emission factor because the utility does not differentiate its emissions by region. However, the facilities included in this study are all located in Northern Saskatchewan, which is powered exclusively by hydropower. Application of a regional emission factor reduces the production-weighted average life cycle GHG emission intensity to 26 kg CO_2e/kg U_3O_8 with a 95% confidence interval of 25 to 29 kg CO_2e/kg U_3O_8. This represents a 42% reduction in life cycle GHG emission intensity from the base case.

Due to the high uranium ore grades found in SK uranium deposits, life cycle GHG emissions intensity for uranium from SK is among the lowest in the world. Further, the life cycle GHG emission intensity estimate from uranium mining-milling in SK is a small (approximately 10%) contributor to the life cycle GHG emissions intensity from the nuclear fuel cycle for light water reactors overall, amounting to approximately 1.2 g CO_2e/kWh electricity (0.6 g CO2e/kWh electricity calculated using the regional hydroelectric power source).

Acknowledgements

This work could not have been achieved without the considerable time, effort, and expertise provided by a number of individuals.

My supervisors, Dr. Gordon Sparks and Dr. Cameron S. McNaughton, have offered invaluable mentorship and guidance throughout the project. Dr. Sparks has encouraged me to consider the big picture while Dr. McNaughton was always willing to sit down for a drink and grind through the details. Thank you both for providing me this opportunity and helping me to get the most out of the MSc program.

The precision and accuracy of this study's results derive from data provided by our industry partners, Cameco Corporation and AREVA Resources Canada Inc. These companies have not only provided a combined 32 years' of production data for the thesis, but have also offered insight into their industries through project meetings, site tours, and frequent correspondence.

Special thanks to Sarah Benson and Dale Huffman at AREVA who have put time and effort into meeting, gathering data, and organizing a site tour, and also to Greg Hesterman who took time out of his schedule to tour Dr. McNaughton and I through the McClean Lake site.

Thank you to Kirsten Ketilson at Cameco who not only provided information and guidance relating to the project, but also took an active interest in my academic and professional development. Thanks as well to Helton Fan, Candice Wist (I think I still owe you donuts), and the small army of individuals who toured Dr. McNaughton and myself through Cameco's facilities.

To my graduate committee, Terry Fonstad and Tom Allen, and to my external examiner, David Sanscartier, thank you for asking all of the tough questions and keeping me on my toes.

Financial help for this project has been generously provided by the Sylvia Fedoruk Canadian Centre for Nuclear Innovation, Innovation Saskatchewan, and the University of Saskatchewan through the Dean's Scholarship, Saskatchewan Innovation and Opportunity Scholarship, Graduate Research Fellowship, and other department scholarships. This funding has not only enabled this project, but has allowed me to pursue training opportunities in life cycle analysis and GHG quantification outside of the university, and to attend and present at conferences.

Finally, I would like to acknowledge my wife Robyn, who did everything she could to support the work, and my young son Elliott, who did everything he could to delay or destroy it. *Don't delete my thesis!*

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List of Abbreviations

BWR	Boiling water reactor
GHG	Greenhouse Gas
g CO_2e	grams carbon dioxide equivalents
GWP	global warming potential
HWR	Heavy Water Reactor
ISL	In situ leaching
ISR	In situ recovery (alternate name for ISL)
ISO	International Organization for Standardization
LCA	Life Cycle Analysis, or Life Cycle Assessment
LWR	Light Water Reactor
kWh	Kilowatt-hour
PWR	Pressurized water reactor
SMR	Small modular reactor
U_3O_8	Uranium oxide or yellowcake

1. Introduction

1.1 Background

As the threat of climate change becomes more urgent, governments are under increasing pressure to enact policies to prevent or, at least, reduce further anthropogenic contributions to climate change by reducing greenhouse gas (GHG) emissions to the atmosphere.

The largest global anthropogenic contributor to GHG emissions is the energy sector, primarily from the combustion of coal, natural gas, and oil used to produce electricity and heat (EPA, 2014). As the largest emitter, this sector is a key target for emission reduction policy. One of the most direct ways to accomplish these reductions is to shift away from heavier emitters such as coal, natural gas, and oil towards lighter emitters such as solar, wind, geothermal, hydroelectric, biomass, and nuclear.

Currently, nuclear power supplies 11% of the world's electricity (World Nuclear Association, 2015). A survey of GHG emission intensity studies of the nuclear power life cycle showed a wide range of values: from 1.4-288 grams carbon dioxide equivalents per kilowatt hour of electricity generated (g CO_2e/kWh) with a mean of 66 g CO_2e/kWh (compared to 800-1700 g CO_2e/kWh for coal) (Sovacool, 2008). Some of the biggest discrepancies in these studies occur when considering the mining-milling of uranium ore which can make up as much as 30% of the total fuel cycle emissions by some estimates (Sovacool, 2008; Warner and Heath, 2012).

Studies reporting GHG emissions from uranium mining-milling vary in their scope and methodology, as well as in the details and assumptions about the uranium sources. Many assess GHG emissions from energy consumption during operation, but exclude emissions from exploration, construction, decommissioning, and/or emissions that are embodied in materials and reagents. Furthermore, there is a lack of recent comprehensive life cycle GHG emission estimates from uranium mining-milling in Saskatchewan (SK), a province that has supplied over 18% of global uranium from 2006-2013 (World Nuclear Association, 2014a).

Saskatchewan is home to exceptionally high-grade uranium ore, averaging up to 18% U_3O_8 in some of the deposits currently being mined (Cameco Corporation, 2013a). Previous studies on the GHG emissions intensity from the nuclear fuel cycle have estimated GHG emissions from global uranium mining-milling operations, of which ore grades typically range from 0.1-0.2% U_3O_8 but may vary to less than 0.05% U_3O_8 (Sovacool, 2008; Warner and Heath, 2012). Yellowcake from higher grade uranium ore such as that found in SK is likely associated with lower life cycle GHG emissions intensity than for other uranium mine-mill facilities around the world.

1.2 Research Objectives

The purpose of this research is to determine the life cycle GHG emissions, expressed in mass of carbon dioxide equivalents (kg CO_2e), created as a result of mining-milling uranium ore in SK to produce yellowcake (kg U_3O_8). Results are expressed as emissions intensity in kg CO_2e/kg U_3O_8. Goals of the research include the following:

1) To quantify life cycle GHG emissions intensity for uranium from SK in terms of kg CO_2e/kg U_3O_8;
2) To identify emissions intensive processes within the uranium mining-milling life cycle; and
3) To identify the largest sources of uncertainty with respect to calculating life cycle GHG emissions inventories for the mining-milling of uranium in SK.

1.3 Scope

1.3.1 System Boundary

This study takes a process-based approach to the life cycle analysis of GHG emissions, which requires that all phases of the product life cycle be taken into account (Weisser, 2007). In the case of the nuclear fuel cycle, this includes everything from construction and commissioning of the mines and nuclear plants through the mining-milling operation, conversion, enrichment, transportation, power generation, facility decommissioning, and waste disposal. This is known as a 'cradle-to-grave' analysis (ISO 14040, 2006).

Since the goal of the study is to determine life cycle GHG emissions intensity per kg U_3O_8 mined and subsequently milled, emissions from the remainder of the nuclear fuel cycle is beyond the scope of this study. Therefore, a 'cradle-to-gate' approach is used. Included in the study are all emissions associated with mine-mill construction, operation, and decommissioning, and from fuel and electricity consumption at corporate headquarters and during uranium exploration. The analysis ends at the 'gate' which is defined at the point where packaged yellowcake product leaves the milling facility. This is illustrated in Figure 1-1.

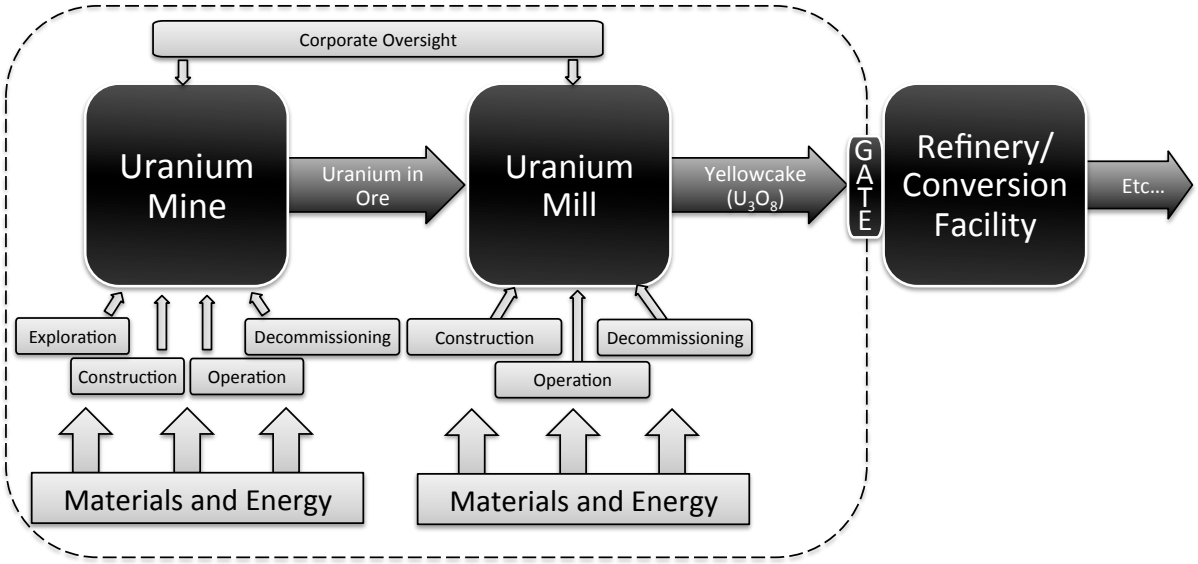

Figure 1-1. 'Cradle-to-Gate' Boundary. Dashed line denotes scope boundary.

1.3.2 Facilities Included

In Saskatchewan, Cameco Corporation (Cameco) and AREVA Resources Canada Inc. (AREVA) are the only uranium mine-mill operators. As project collaborators, these companies have supplied data on GHG emissions and emissions-related processes for the following facilities:

- McArthur River Operation-Key Lake Operation mine-mill;
- Eagle Point-Rabbit Lake Operation mine-mill; and
- McClean Lake Operation mine-mill

Only these facilities are considered for detailed life cycle analysis. Together, they represent 97% of Canada's uranium production from 2000 to 2013 (World Nuclear Association, 2014b).

In addition to the above-mentioned facilities, AREVA has provided details for the decommissioning of Cluff Lake Project, an operation that included underground and open pit uranium mines as well as a uranium mill. Mining-milling at Cluff Lake ended in 2002 and decommissioning activities began soon after. In 2013, the last buildings were demolished and site occupancy ceased. Active decommissioning is now complete and the site is in a period of long-term monitoring (AREVA Resources Canada Inc., 2014c). While Cluff Lake Project is not included for detailed analysis, the recent decommissioning activities are used to help validate estimates for the other facilities.

1.3.3 Emission Sources

In each phase of the life cycle, the study considers direct GHG emissions, energy indirect emissions, and other indirect emissions as defined by ISO 14064:2006. These emission categories are commonly referred to as Scope 1, 2, and 3 respectively (WRI/WBCSD, 2014a). Scope 1 emissions come directly from company-owned vehicles, equipment, and facilities (ISO 14064, 2006). Scope 2 includes emissions

generated from purchased energy such as electricity or steam (ISO 14064, 2006). Scope 3 emissions are indirect, and typically include mobile emissions from employee transport and embodied emissions in equipment and building materials (ISO 14064, 2006).

The number of emissions sources included in Scope 3 can be almost unlimited, so it is necessary to establish cut-off criteria. Cut-off criteria specifies "the level of environmental significance associated with unit processes or product system to be excluded from a study" (ISO 14044, 2006). In this study, the cutoff criteria is set at 0.1% of the life cycle GHG emissions for each mine-mill although many processes are included that do not meet this criteria. In many cases, a generic life cycle inventory database is consulted as a means to include these smaller contributors.

ISO 14040:2006 does not provide specific guidance on how to set the cut-off criteria, but states that exclusions are only permitted if the omission does not change the overall conclusions of the study. An alternate life cycle GHG assessment standard, *Publically Available Specification (PAS) 2050:2011* suggests a significance threshold of 1% for any given unit process and requires that at least 95% of the complete product life be included (British Standards Institution, 2011).

1.4 Conceptual Methodology

Two pieces of information are required to calculate GHG emissions: an activity factor and an emission factor. The activity factor describes a particular process (e.g., burning of propane) and the emission factor quantifies the emissions associated with that process (e.g., 1.95 kg CO_2/L of propane). The overall emissions are calculated by multiplying the activity factor and the emission factor. Emissions intensity is calculated by dividing the total emissions (kg CO_2e) by total production (kg U_3O_8).

Data on emissions-relevant activities are provided by the project collaborators: Cameco and AREVA. Supplementary information is extracted from publicly available sources (e.g. mining companies' websites, regulatory reports).

Emission factors are available from a number of sources, predominantly Environment Canada (fuel burning), SaskPower (electricity consumption), the ecoinvent v3.0 life cycle inventory database, and other peer-reviewed literature. Emission factors are discussed in more detail in Section 3.3.1.

A full discussion of methodology is presented in Section 3.

1.5 Organization of Thesis

This thesis is presented in the traditional format. The literature review (Chapter 2) outlines globally accepted methodologies used in calculating and reporting on GHG emissions and outlines several life cycle analysis approaches. This is followed by an overview of the entire nuclear fuel cycle for light water reactors, with some discussion of the differences in the fuel cycle of heavy water (e.g., Canadian Deuterium-Uranium; CANDU) and Generation IV reactors.

Section 2.5 is a comprehensive literature review of GHG emissions from the entire nuclear fuel cycle as reported in the current literature. It includes a review of three meta-analyses and the studies on which they are based. Where available, emissions are reported by life cycle phase (e.g., mining-milling, enrichment, etc.). Also included are the results of a seminal study undertaken by Warner and Heath (2012). This review considered 274 nuclear studies, eventually harmonizing then evaluating the results from 99 independent estimates of emissions from light water reactor-based nuclear life cycles. Although the present study considers only the mining-milling phase of the nuclear life cycle, it is important to understand the context in which these activities occur, hence the inclusion of the full fuel cycle studies.

Chapter 3 begins by discussing the uranium mining-milling operations, past and present, in northern SK, and introduces the facilities included in this study. The chapter goes on to detail the life cycle analysis methodology, data collection methods, and modeling approach employed by this study and also describes the approach used to incorporate and evaluate uncertainty in this study's emissions estimates.

Chapters 4 and 5 present the results and conclusions of the study respectively.

2. Greenhouse Gas Emissions, Life Cycle Analysis, and the Nuclear Fuel Cycle

To establish the context of this research, the following literature review discusses the relationship between the emissions of greenhouse gases (GHGs) and climate change; describes the life cycle approach to accounting for environmental impacts in general (ISO 14040); and details ISO standards specific to GHG accounting (ISO 14064).

Section 2.4 describes the full nuclear life cycle and is followed by a review of studies that estimate GHGs from both the full nuclear life cycle and the mining-milling phases specifically.

To situate nuclear power emissions intensity in the context of other electricity-generating technologies, the literature review concludes with a discussion of GHG emissions from non-nuclear power sources.

2.1 Climate Change and GHG Global Warming Potentials

The relative contributions of the driving forces of climate change are quantified by radiative forcing (RF), "the net change in the energy balance of the Earth system due to some imposed perturbation", typically expressed in watts per square meter averaged over some defined time and space (Myhre et al., 2013). Simply put, radiative forcing quantifies the change in electromagnetic energy entering and leaving the Earth's atmosphere due to an imposed change. It is not an attempt at representing overall climatic response but the relative influences of different factors.

In the case of human-driven climate change, RF is partly a result of GHGs entering the atmosphere due to human activities. GHGs and other pollutants have a complex effect on the climate. Some contribute to warming by trapping outgoing electromagnetic radiation, while others cool the planet. The interaction of different elements at different altitudes creates complex effects that are not fully understood at this time (Myhre et al., 2013).

However, the effects of increasing atmospheric concentration of some of the most prominent gases are well understood (Kaplan, 1960; Myhre et al., 2013; Plass, 1956). Of these gases, RF is dominated by carbon dioxide (CO_2 ~64%), methane (CH_4 ~17%), and nitrous oxide (N_2O ~6%) (Myhre et al., 2013). The magnitude of their effect is due to their total concentration in the atmosphere, each chemical's specific capacity to trap heat, and each chemical's residence time in the atmosphere. For example, methane has 28-86 times the heat trapping capacity of carbon dioxide, but is 217 times less prominent and has a shorter residence time (Myhre et al., 2013). Therefore, net RF from methane is less than from carbon dioxide.

For the sake of analysis, the heat-trapping capacities of different GHGs are normalized to the warming potential of carbon dioxide on a mass basis. The resultant factor is a gas-specific global warming potential (GWP). GWPs for major GHGs as reported in the industry-standard Intergovernmental Panel of Climate Change Fifth Assessment Report, are listed in Table 2-1.

Table 2-1. 100-Year Global Warming Potentials (GWP) of Major Greenhouse Gases without Climate-Carbon Feedbacks from Intergovernmental Panel on Climate Change Fifth Assessment Report (Myhre et al., 2013)

Greenhouse Gas	GWP_{100}
CO_2	1
CH_4	34
HFC-134a	1550
CFC-11	5350
N_2O	298
CF_4	7350

The total emissions from a particular process are calculated by:

$$\sum_i \boldsymbol{Mass}_i \times \boldsymbol{GWP}_i \qquad \textbf{(2.1)}$$

where *i* cycles through each GHG emitted. Total emissions are expressed in kilograms of carbon dioxide equivalents (kg CO_2e) or similar. In this way, processes can be directly compared on the basis of their contribution to climate change.

2.2 Life Cycle Analysis - ISO 14040:2006

Life cycle analysis (life cycle assessment, LCA) is an environmental management technique used to assess environmental impacts of various products (including services) and can be extended to make comparative assertions between similar products (ISO 14040, 2006). Examples of 'products' that could be evaluated using LCA include electrical energy, T-shirts, vehicles, and myriad others (ISO 14040, 2006).

To make comparative assertions that are fair and accurate, a standard approach to LCA is necessary. The ISO 14040:2006 standard provides a framework within which to conducts LCA studies.

The ISO 14040:2006 standard describes four phases of the LCA:

1. Goal and Scope Definition
2. Life cycle inventory analysis (LCI) phase
3. Life cycle impact assessment (LCIA) phase
4. Interpretation phase

LCA is an iterative technique. As such, output from each phase informs not only later phases, but also previous ones. This is illustrated in Figure 2-1.

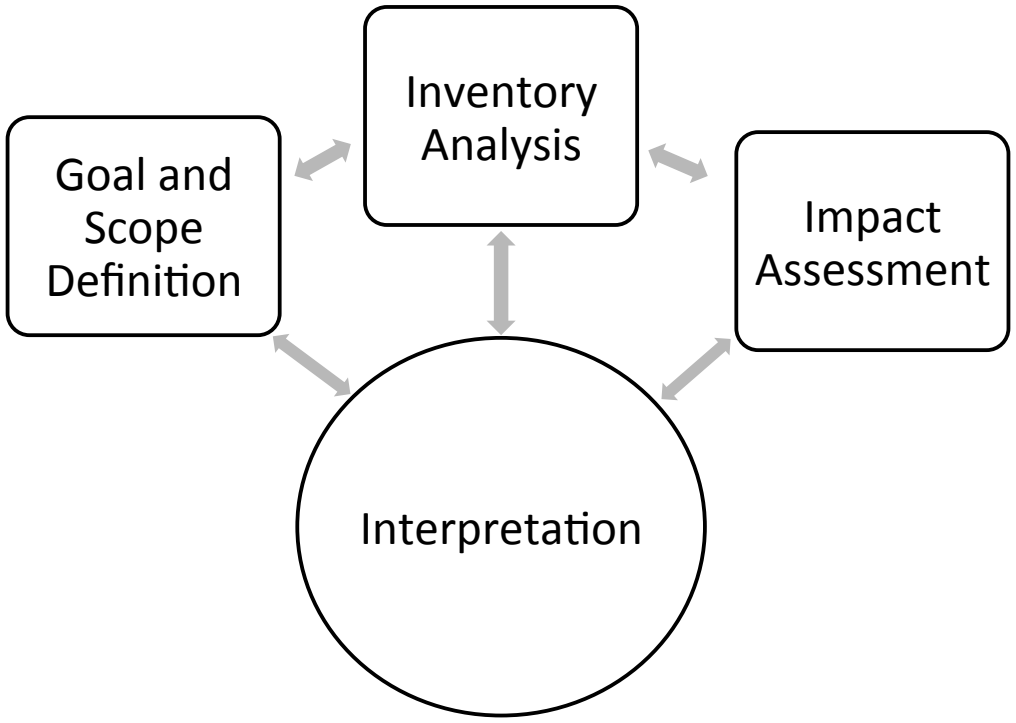

Figure 2-1. Phases of Life cycle Analysis, redrawn from ISO 14040 (2006)

2.2.1 Phase 1: Goal and Scope Definition

The goal of the LCA study should be explicitly stated. Its definition includes the intended application, the reason for carrying out the study, and the intended audience (ISO 14044, 2006). If the results are going to be disclosed to the public or used for comparative analysis, these intentions should be included in the goal definition (ISO 14044, 2006).

The LCA scope describes the depth and breadth of the study, and typically incorporates the following elements (ISO 14044, 2006):

- Product systems to be studied;
- Functions of product system;
- Functional unit;
- System boundary;
- Allocation procedures;
- Life cycle impact assessment (LCIA) methodology and types of impacts;
- Interpretation to be used;
- Data requirements;
- Assumptions;
- Value choices and optional elements;
- Limitations;
- Data quality requirements;
- Type of critical review, if any; and
- Type and format of the report required for the study.

A few of these terms require further explanation.

A **'functional unit'** is the basic unit of analysis in the LCA and should be clearly defined and measureable (ISO 14040, 2006). Examples include: 1 kWh of electrical energy leaving power plant, 1 T-shirt, or 1 electric vehicle.

All impacts are related back to this functional unit. For example, there may be 20 g CO_2e emitted for every kWh of electrical energy leaving a nuclear power plant. These emissions come partly from operation of the plant but also from upstream and downstream processes.

In an LCA, a **'unit process'** is the smallest sub-process with defined inputs and outputs (ISO 14044, 2006). These processes are linked to each other by flows of intermediate products and to the environment by elementary flows. These '**elementary flows'** include material or energy drawn from or emitted to the environment without additional human transformation (ISO 14044, 2006).

The **'system boundary'** defines what unit processes will be included in the study. Ideally, the model will be set up in such a way that all inputs and outputs at the systems boundary are elementary flows (ISO 14040, 2006).

Including all elementary flows can be very resource-intensive. A large effort may be required to include processes that have relatively little impact on the results and/or little-to-no impact on recommendations resulting from the study. For this reason, cut-off criteria can be used in place of exhaustive analysis. The basis for exclusion

should be clearly stated (ISO 14044, 2006). Typically, items are excluded that cumulatively contribute less than a defined percentage of the total mass, energy, or environmental impacts of a system (ISO 14040, 2006). In this study, the cutoff criteria is set at 0.1% of the life cycle GHG emissions for each mine-mill although many processes are included that do not meet this criteria.

The quality of an LCA study is critically dependent on the definition of the system boundary. In comparative analysis, it is imperative that boundaries are consistent between studies (ISO 14040, 2006).

In general, life cycle stages that should be considered include (quoting (ISO 14040, 2006)):

- Acquisition of raw materials;
- Inputs and outputs in the main manufacturing/processing sequence;
- Distribution/transportation;
- Production and use of fuels, electricity and heat;
- Use and maintenance of products;
- Disposal of process wastes and products;
- Recovery of used products (including reuse, recycling and energy recovery);
- Manufacture of ancillary materials;
- Manufacture, maintenance and decommissioning of capital equipment; and
- Additional operations, such as lighting and heating.

A consideration of all life cycle stages ensures that environmental burdens are not shifted between stages to bias the perceived environmental performance of a system (ISO 14040, 2006).

2.2.2 Phase 2: Life Cycle Inventory Analysis (LCI)

The second phase of analysis is model construction. Data is collected for all inputs and outputs within the system boundary. These data are related to the appropriate unit processes which are, in turn, related to the functional unit (ISO 14044, 2006). Data should be validated at multiple stages during this phase. This could be achieved by mass balance, energy balance, or other methods (ISO 14044, 2006).

As data is collected, processed, and interpreted, the LCA scope and system boundary may need to be adjusted. This will require the collection of more data as the LCI process repeats (ISO 14044, 2006).

In reporting, data sources should be referenced along with data quality indicators. Unit processes should be explicitly described and calculations should be included. This ensures adequate transparency which allows the reliability of the results to be scrutinized and accurate comparisons to be made (ISO 14044, 2006).

2.2.3 Phase 3: Life Cycle Impact Assessment (LCIA)

In the third phase of the LCA, impacts are quantified. These are usually environmental impacts, but can include social and economic impacts as well. There are three mandatory elements in the LCIA phase (ISO 14044, 2006):

1. Selection of:
    a. Impact categories;
    b. Impact category indicators; and
    c. Characterization models.
2. Classification:
    a. Assignment of LCI results to impact categories.
3. Characterization:
    a. Calculation of category indicator results.

The impact categories selected should reflect the breadth of environmental issues associated with the system under consideration (ISO 14044, 2006). Many impact categories are pre-defined, but new categories can be created if required by the specific goals and scope of the study (ISO 14044, 2006).

The example impact category given in ISO 14044:2006 is 'acidification'. The relevant LCI results include NO_x and SO_2 emissions, and the category indicator is proton release (H^+ aq).

One of the most relevant impact categories in power generation is 'climate change'. The LCI results would be mass of carbon dioxide GHG-equivalents released (g CO_2e), per functional unit (e.g., kWh electrical energy produced). The characterization model is based on the Intergovernmental Panel on Climate Change Fifth Assessment Report model (Myhre et al., 2013), and the 100-yr global warming potential for each gas is chosen as the characterization factor. The units of the category indicator results might be g CO_2e/kWh (ISO 14044, 2006).

2.2.4 Phase 4: Interpretation

Interpretation occurs throughout the LCA process. As more information is considered, new issues are discovered and, in response, the goals and scope of the study are refined, unit processes are added or modified, the functional unit can change, and impact categories are added or rejected (ISO 14044, 2006).

Interpretation includes evaluating the items in Table 2-2 (ISO 14044, 2006).

Table 2-2. Items Evaluated in Interpretation Phase of LCA

Completeness	Are all relevant data required for interpretation included?
Sensitivity	How are the final results and conclusions affected by uncertainties in the data, allocation methods, etc.?
Consistency	Are the assumptions, methods, and data consistent with the goal and scope?

2.2.5 Reporting and Critical Review

The results of the LCA along with the data, methodology, assumptions, and limitations should be objectively reported in a complete and transparent way (ISO 14044, 2006).

If the results are to be used in comparative assertions and disclosed to the public, they should be subjected to a formal critical review conducted by an LCA expert independent of the LCA under review (ISO 14044, 2006). This is to ensure (quoting ISO 14044:2006):

- The methods used to carry out the LCA are consistent with ISO 14044:2006;
- The methods used to carry out the LCA are scientifically and technically valid;
- The data used are appropriate and reasonable in relation to the goal of the study;
- The interpretations reflect the limitations identified and the goal of the study; and
- The study report is transparent and consistent.

2.2.6 LCA Methodology

ISO 14040:2006 allows for a variety of methodological approaches to suit the specific needs and goals of the study being performed. A few standard methodologies have emerged, each with strengths and limitations. A few of these are discussed in the following sections. For this study, the more rigorous Process Chain Analysis methodology has been chosen.

2.2.6.1 Process Chain Analysis (PCA)

Perhaps the most rigorous LCA approach, process chain analysis (PCA) is a vertical bottom-up technique (Weisser, 2007). When considering a particular product, PCA examines all of the materials involved and the unit processes upstream and downstream of the product system. This requires a detailed knowledge of all processes in a system and a complete bill of materials. Additionally, manufacturing information for each material is required (Weisser, 2007).

Process chain analysis has the advantage of being specific and accurate and makes the identification of 'hot spots' in the process chain straightforward (Weisser, 2007). On the other hand, it is very resource-intensive to achieve an appropriate level of detail and it is not always possible to obtain sufficient data for all relevant products and processes (Weisser, 2007). The ideal LCA performed this way would necessarily consider the entire economy, an impossible undertaking. Therefore, process chain analysis involves a systematic underestimation of results due to truncation at the system boundary (Weisser, 2007). The use of a pre-existing life cycle inventory database can allow for the inclusion of additional processes and an expansion of the system boundary to help minimize the problem of underestimation. In this study, the ecoinvent database is used for this purpose (ecoinvent Centre, 2013).

2.2.6.2 Economic Input/Output Analysis (EIO)

An alternative to process chain analysis is economic input/output analysis (EIO), a statistical top-down approach (Weisser, 2007). Here, a product is divided into its economical components: machinery, chemistry, services, etc. (Raadal et al., 2011). Emissions of each component are then calculated based on the economic value of each multiplied by the environmental performance of the associated economic sector (Raadal et al., 2011).

EIO is considerably less resource-intensive than PCA and allows for the inclusion of more emissions by avoiding system boundary truncation errors. However, EIO has some severe limitations (Fthenakis and Kim, 2007). When the process chain does not conform to industry averages, significant errors can occur in emission estimates. For example, Fthenakis and Kim (2007) found that GHG emissions in the construction of nuclear plants were estimated to be 10-20 times higher when using EIO compared to PCA. The authors argue that the costs of materials increase appreciably due to increased safety standards, but that there is not an equivalent increase in associated emissions.

2.2.6.3 Hybrid Approach (H-IOA)

To combine the detail of PCA and inclusiveness of EIO, a hybrid analysis (H-EIO) is sometimes used. In this approach, the most influential processes are modeled using PCA while less important, secondary processes are modeled using EIO (Weisser, 2007).

In a simplified version of the H-IOA approach, the overall monetary cost is multiplied by the economy-wide energy intensity. This method is called average energy intensity (AEI) (Beerten et al., 2009). AEI is appealing due to its simplicity, but has been shown to result in unacceptably large errors (Beerten et al., 2009).

2.3 GHG Inventory Quantification - ISO 14064:2006

Life cycle analysis is used to quantify a number of different environmental impacts, including GHG emissions. The ISO 14064:2006 standard outlines principles and requirements for developing organization-level GHG inventories. While it does not specify a life cycle approach, some of the ISO 14064:2006 principles can be useful to apply, especially in helping organizations to understand and improve GHG management.

GHG emissions are calculated based on elements within a specified organizational boundary. This boundary may include more than one facility. In some cases, multiple organizations may have an interest in a single facility. In this event, GHG emission can be allocated in one of they following ways (ISO 14064, 2006):

- **Control**: organization that has operational or financial control accounts for GHG emissions/removals; or
- **Equity Share**: organization accounts for its portion of GHG emissions based on its equity share in the facility.

The allocation method is a matter of choice but should be consistent throughout the GHG emissions quantification project (ISO 14064-1, 2006).

Within the defined boundaries, GHG emissions are divided into direct, energy indirect, and indirect emissions categories. These emission categories are commonly referred to as Scope 1, 2, and 3 respectively (WRI/WBCSD, 2014a; ISO 14064, 2006). These classifications are defined as follows (ISO 14064, 2006):

- Scope 1: Direct GHG Emissions
    - Electricity, heat, and steam generation within organization boundary
        - Not from biomass
    - Combustion of fuel in company owned vehicles
    - Emissions from industrial processes
- Scope 2: Energy Indirect Emissions
    - Emissions from generation of imported electricity, heat, or steam consumed by the organization
- Scope 3: Other Indirect GHG Emissions
    - Commuting and business travel by employees
    - Transportation of an organization's products, materials, people, or waste by another organization
    - Outsourced activities
        - e.g. waste management
    - Emissions from use and end-of-life phases or organization's products and services
    - Emissions from production of purchased raw or primary materials
    - Other

From this description, it is clear that a life cycle approach to GHG inventory quantification will need to include Scope 3 emissions.

ISO 14064:2006-1 describes the following five steps in the quantification of GHG emissions:

1. Identification of GHG sources and sinks;
2. Selection of quantification methodology;
    a. Calculations based on models, correlations, mass balances;
    b. Continuous or intermittent measurement; or
    c. A combination of calculation and measurement;
3. Selection and collection of GHG activity data;
4. Selection or development of GHG emission factors; and
5. Calculation of GHG emissions/removals.

GHG emissions inventories should be assessed annually and compared against a base year. In this way, performance can be tracked and reported and appropriate emission mitigation strategies enacted (ISO 14064, 2006). To be consistent with the standard, results that are to be publicly reported should be subjected to verification by an approved third party examiner (ISO 14064, 2006).

2.4 The Nuclear Fuel Cycle

The following section is an overview of the nuclear fuel cycle from resource extraction through to waste management.

2.4.1 Base Case - Light Water Reactors

Over 80% of the reactors in commercial operations today are light water reactors (LWR), most of which are Generation II pressurized water reactors (PWR), followed in popularity by Generation II boiling water reactors (BWR) (World Nuclear Association, 2014c). As such, most available data is based on these types of reactors and the present discussion is centered on them. At the end of this section, other current and future generating technologies are discussed.

The nuclear fuel cycle is illustrated in Figure 2-2.

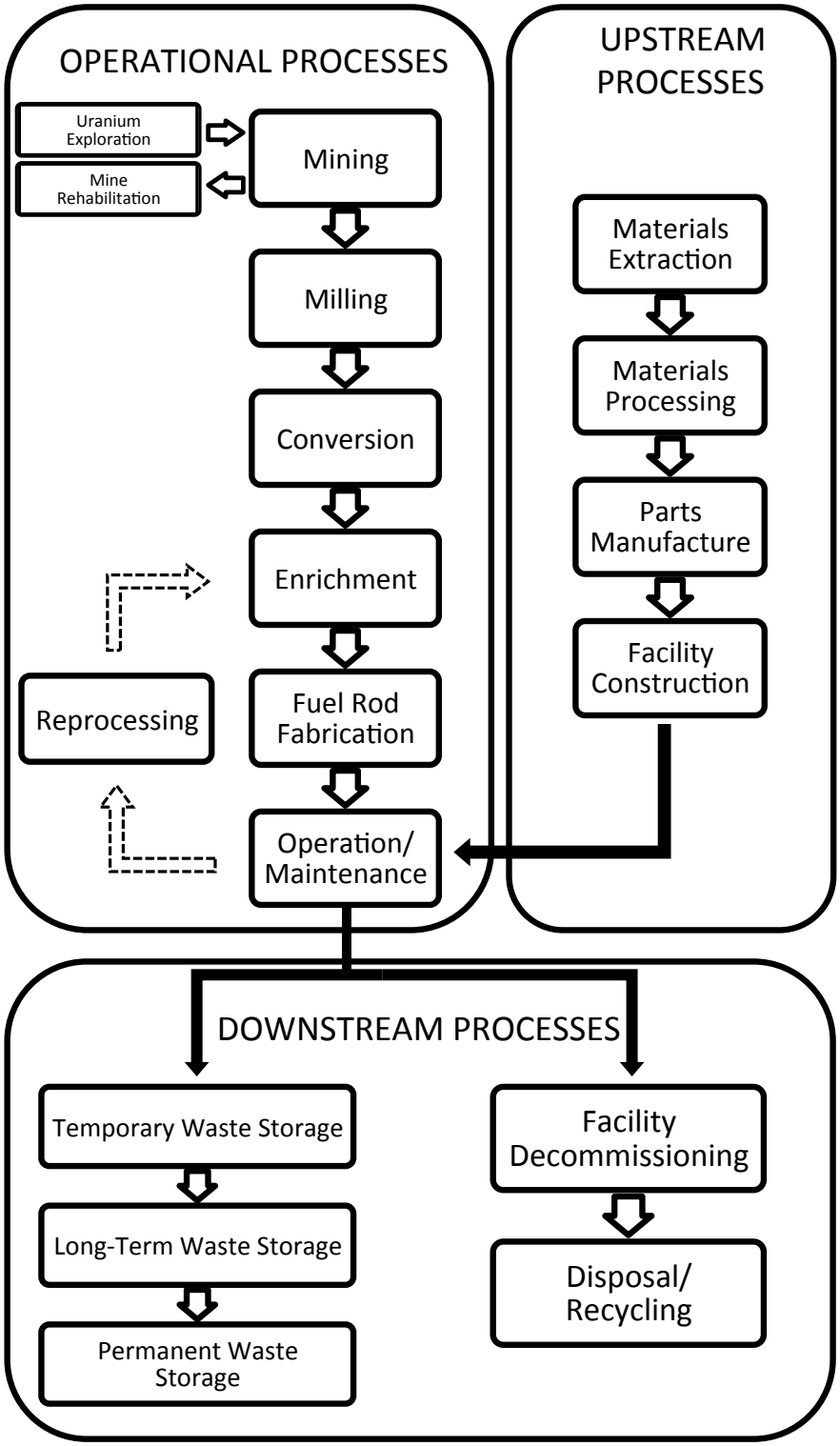

Figure 2-2. Schematic of Nuclear Fuel Cycle. Redrawn from Warner and Heath (2012). Reproduced with permission from the publisher.

The main processes in each stage are discussed in the following sections.

2.4.1.1 Exploration

The nuclear fuel cycle begins with exploration for uranium-rich ores. The radioactive nature of uranium ore makes it detectable via radiometric surveys. Barretto (1981) identifies several stages of exploration:

1. Reconnaissance stage
    a. Potential sites identified based on large-scale geology
2. Follow-up stage
    a. Aerial radiometric surveys
        i. Sensitive detectors mounted on helicopters or fixed-wing aircrafts used to evaluate large areas
    b. Geochemical surveys of stream sediments, water, soil, and soil-gas
3. Detailed stage
    a. Surface radiometric survey
        i. Ground-based follow-up using portable detectors and borehole loggers
    b. Geological mapping, prospecting, trenching
4. Exploration development
    a. Detailed mapping, trenching
    b. Close-grid drilling
    c. Mineralogical and petrographic study

The decision to move forward from one stage to the next is dependent on the evidence of uranium-rich ores, ore grades, cost of recovery, market demand, the regulatory environment, and other factors.

2.4.1.2 Mining

Once a uranium ore-body is discovered, engineering studies show that mining is technically and economically feasible, and regulatory approval is granted, a mine can be established. The most common types of mines are underground and open pit, although in situ leaching (ISL, in situ recovery, solution mining) can be used in specific circumstances (World Nuclear Association, 2014b; World Nuclear Association, 2013).

Open pit mining is generally employed when ore bodies are at or close to the surface. When the ore body is located at depth, overburden must be removed and the overlying rock is removed and stockpiled as waste rock (World Nuclear Association, 2012). The suitability of open pit mining for uranium extraction is limited by the ore-stripping ratio: the ratio of the volume of overburden or other waste materials handled to extract some volume of ore. This is a function of the deposit's depth (IAEA, 2000).

Open pit mines usually use conventional pit designs and equipment with some special provisions made to protect workers and the surrounding environment from excessive radiation exposure (IAEA, 2000). As ore grade increases beyond 1% U_3O_8, the level of safeguards required increases greatly. These can include the incorporation of shielding devices on loading and hauling equipment to protect workers from gamma radiation; air monitoring for radon and other airborne contaminants; control of surface and underground water flow; and isolation of radioactive waste rock from the environment (IAEA, 2000). Some examples of open pit uranium mines are shown in Figures 2-3, and 2-4.

Figure 2-3. Open Pit at Rössing Uranium Mine, Namibia. Reused under license. (Ikiwaner, 2009)

Figure 2-4. Mined-out and Flooded Open Pits at McClean Lake Operation, Saskatchewan

In some situations, the extraction of low-grade ore can be made economical by co-mining. In co-mining, multiple resources are extracted from a single mine and sorted in a downstream process (IAEA, 2000).

Underground mining is employed when ore deposits are deeper. This approach can potentially have lower environmental impacts due to a smaller surface footprint (World Nuclear Association, 2012). Access to the ore body is via access shafts and tunnels (World Nuclear Association, 2012). To the extent possible, conventional underground mining methods are used. Mining methods are often modified to address the particular safety concerns present in a radioactive environment (IAEA, 2000). As with open pit mining, increasing ore grades necessitate greater precaution. In some cases, mining equipment is remotely operated to avoid human exposure to significant radiation dosing (IAEA, 2000).

Large ventilation systems supply fresh air to underground mines to keep radon and other airborne radioactive contaminants at safe levels (IAEA, 2000). In cold climates, large heaters are required to heat the incoming air in order to prevent freezing of mining equipment (IAEA, 2000).

The control of water is a major concern in all types of uranium mines, but is of particular concern in underground uranium mining. Radon is dissolved in subsurface waters under pressure. As this water enters the tunnels and is exposed to atmospheric pressure, the dissolved radon is released to the air and quickly concentrates (IAEA, 2000). Moreover, if water is present in sufficient quantities, it can cause instability in the surrounding rock.

Water control measures include sealing tunnel walls with a cement product and/or collecting water in special drainage tunnels below the operations area (IAEA, 2000). In some cases, the area in the vicinity of the ore body is frozen to ensure tunnel wall stability and prevent water flow (IAEA, 2000). Figure 2-5 shows some of the infrastructure required to maintain an underground freeze wall at McArthur River Operation underground mine.

Figure 2-5. Freeze Wall Infrastructure at McArthur River Operation

When ore bodies lie in unconsolidated material such as sand or gravel, ISL can be used (World Nuclear Association, 2012). However, this technology has fairly specific geological requirements. The formation is, ideally, a high permeability horizontal aquifer confined by low permeability rock from above and below (World Nuclear Association, 2012; IAEA, 2000). This extraction technique involves pumping an oxygen-rich, weakly acidic solution through the aquifer to dissolve the uranium. The pregnant solution is then pumped out of the aquifer and the uranium is recovered as a precipitate at a surface treatment facility (World Nuclear Association, 2012). ISL can allow for economical recovery of low ore grades and decreased human and environmental exposure to hazards (IAEA, 2000). Once all of the economically recoverable uranium has been extracted, the aquifer is normally required to be replenished. This can take several years and represents a large fraction of the total cost (IAEA, 2000).

2.4.1.3 Milling

Once mined, uranium ore is transported to a mill. These facilities are usually located near the mine to reduce costs. Once at the mill, the ore enters the process circuit. A simplified schematic of the uranium milling circuit is illustrated in Figure 2-6.

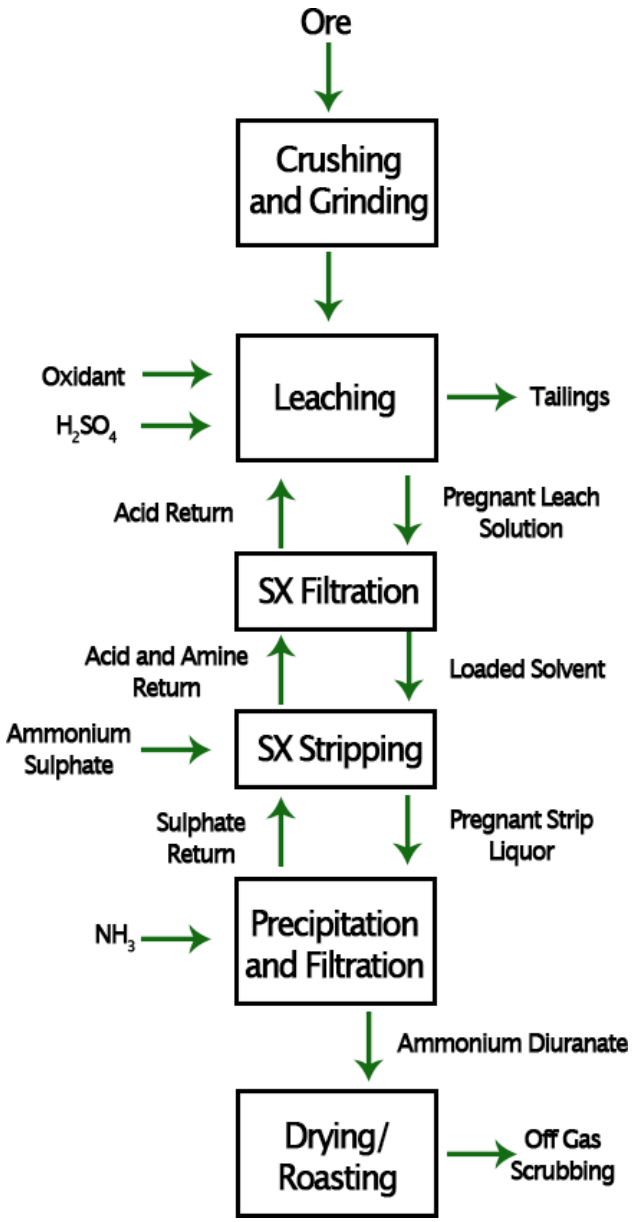

Figure 2-6. Uranium Mill Process Diagram, redrawn from (World Nuclear Association, 2012). SX indicates solvent extraction. This figure is reproduced with permission from the publisher.

First, the ore is weighed and sampled for moisture content. It is then fed into the process circuit on a dry weight basis (IAEA, 2000). After crushing and grinding, the ore enters the leaching circuit where it is mixed with acid (usually sulfuric acid) and an oxidizing agent (usually manganese dioxide, sodium chlorate, pyrolusite, oxygen, or hydrogen peroxide) (Edwards and Oliver, 2000). After leaching, the uranium is now dissolved in solution (World Nuclear Association, 2012). The pregnant leach solution is washed and thickened before a solvent exchange (SX) or ion exchange (IX) process is used to extract uranium from the leachate, leaving behind other metals such as vanadium, molybdenum, and iron (IAEA, 2000).

In solvent extraction, the pregnant leach solution is first filtered to remove any remaining solid particles. The pregnant solution enters one end of the circuit while a barren solvent solution is introduced at the other end. The two liquids run countercurrent through several stages of solvent extraction until most of the uranium has been removed from the leach solution. The barren solution is then returned to the leach circuit (IAEA, 2000). The loaded solvent may be treated to remove impurities. Cations are removed by reducing the pH to ~1.5 with the addition of sulfuric acid. Anions are removed by exposing the solvent to gaseous ammonia (World Nuclear Association, 2012).

The next stage is the stripping circuit where loaded solvent is stripped of uranium in multi-phase process using a stripping agent, often ammonium sulfate (World Nuclear Association, 2012; IAEA, 2000). The solute is then precipitated as yellowcake when the solution is neutralized with the addition of gaseous ammonia. The precipitate is then dewatered and roasted to yield the yellowcake product ready for packaging (World Nuclear Association, 2012). The yellowcake product is ~85% uranium metal by weight (Adamantiades and Kessides, 2009).

The previous description is a generic overview of the milling process. There are many variations to this approach. Some mills use an alkaline leaching circuit instead of acid, or a resin/polymer ion exchange process instead of solvent extraction (IAEA, 2000; World Nuclear Association, 2012). Mills also vary in their level of pre-processing. At some SK mines that have very high ore grades, the uranium rich ore is downgraded by mixing it with low grade waste rock before entering the normal milling circuit (Cameco Corporation, 2012a). At other sites, part of the milling process described happens at the mine site. For example, at the McArthur River and Cigar Lake underground mines, ore is crushed and ground underground, then transported to the mill as slurry (Cameco Corporation, 2012a).

When ore grades are low, it may be economical to perform presorting to upgrade the rock's uranium content. There are several approaches to sorting (IAEA, 2000):

1. Radiometric sorting
    a. Ore is crushed to 30-150mm
        i. Separated by size category
        ii. Radiometric scanner measures total radioactivity
        iii. Laser scanner measures horizontal surface of stones
        iv. Ratio of radioactivity/surface calculated by computer and compared to preset cut-off grade
        v. Compressed air jet directed by computer sorts pebbles according to above ratio
2. Magnetic sorting
    a. Removes iron sulfides which can spoil yellow cake
    b. Ceramic magnets used to collect pyrrhotite

3. Gravimetric sorting
    a. Appropriate when uranium is found in pitchblende which has higher specific gravity than waste rock
    b. Sorting is done by centrifuge
4. Grain size classification
    a. Useful when uranium is in particular grain size fraction
5. Flotation
6. Mechanized shredding

The appropriate ore treatment depends on mineralogy of deposit, ore grade, and other factors.

2.4.1.4 Mine and Mill Decommissioning and Tailings Management

Uranium mine and mill decommissioning can be a significant portion of the total cost of production.

IAEA (2000) states that radioactive waste and radiation-contaminated equipment must be somehow isolated from the environment by containment, cleaning, or some other means. In open pit mines, waste rock is often backfilled into the pit which can, in some instances, serve as a repository for mill tailings and contaminated equipment as well. Pits may be allowed to fill with water which helps attenuate the release of radon gas. Water covers are also considered an effective method to reduce acid rock drainage and heavy metal release from tailings (Yanful et al., 2004).

If water covers are used, water quality must be monitored and often requires treatment before release into the environment (World Nuclear Association, 2012; IAEA, 2000). In dry climates, tailings are covered in thick clay caps topped with a protective layer of rock and a layer of topsoil (IAEA, 2000; World Nuclear Association, 2012).

If stored above ground, mill tailings need to be covered with enough material to provide shielding from gamma radiation. Waste rock piles must be contoured to reduce instability, and the area needs to be revegetated. If waste rock contains acid-producing minerals or sufficiently high levels of radiation, additional effort must be made to isolate it from the surrounding environment (IAEA, 2000).

The amount of tailings generated is directly related to ore grade. The processing of lower ore grades yields more tailings in proportion to the decrease in ore grade (Sovacool, 2011).

Tailings repositories require long-term monitoring to ensure continual protection of the surrounding environment (IAEA, 2000).

Underground mines must be backfilled or fitted with permanent engineered caps. As in open pits, spent underground mines can be used as a repository for mill tailings

and contaminated equipment. The hydrology and stability of the region must be modeled to determine what the potential for surface hazards are (IAEA, 2000).

ISL operations are generally easier to decommission due to their small surface operations and minimal waste production. However, the rehabilitation of the aquifer can be very time-consuming and expensive (IAEA, 2000).

2.4.1.5 Refining

Once yellowcake is dried and packaged, it is shipped to a refinery where impurities are removed and the product is converted to UO_3 (Cameco Corporation, 2014b).

In this process, yellowcake is dissolved in nitric acid and goes through another solvent extraction process (Edwards and Oliver, 2000). At this stage, the uranium-rich solution is called OK liquor. Nitric acid and water are boiled off, leaving uranyl nitrate hexahydrate (Cameco Corporation, 2014b). This is fed into denitrification pots where it breaks down chemically into UO_3 and nitrogen oxides (Cameco Corporation, 2014b). The granular UO_3 is removed and stored for transport to a conversion facility (Cameco Corporation, 2014b). While this is Cameco's approach, there are other refining techniques that achieve the same result (Edwards and Oliver, 2000).

2.4.1.6 Conversion

In the conversion stage, uranium's form changes from UO_3 to UF_6. The UF_6 can be generated using fluorine gas at elevated temperature, but is not commercially employed since F_2 is an expensive and hazardous compound (Edwards and Oliver, 2000).

One commercial alternative is to use hydrogen fluoride (HF) in the 'dry fluoride volatility process' (Cameco Corporation, 2014b). In this process, HF is split by electrical current into H_2 and F_2 gases. UO_3 and H_2 are heated in fluid bed reactor whereby UO_3 is reduced to UO_2 in a powder form (Cameco Corporation, 2014b). The powder is mixed with HF in a wet reactor tank to form a UF_4 slurry which is subsequently dried and calcinated, producing UF_6 powder (Cameco Corporation, 2014b). This is fed into a flame reactor with F_2 gas yielding UF_6 gas which is cooled to liquid form, transferred to a storage container, and shipped to an enrichment facility (Edwards and Oliver, 2000; Cameco Corporation, 2014b).

Special care must be taken in handling products and wastes in this process as some of the natural radiation shielding of the radioactive decay products is removed (Edwards and Oliver, 2000).

2.4.1.7 Enrichment

When required as part of the nuclear life cycle, enrichment is one the most energy-intensive processes. The predominant isotope in natural uranium is U-238, which makes up 99.2% of its elemental composition. U-238 however, is not fissile. In the enrichment step, the fissile U-235 isotope content of the fuel, which exists at 0.7% concentration naturally, must be brought up to 3.5% or higher for use as in most commercial light water nuclear reactors (Sovacool, 2011).

Because they use heavy water in place of light water, Canadian Deuterium Uranium (CANDU) reactors do not require U-235 enrichment. This cost and energy trade-off is discussed in Section 2.4.2.

Historically, the two most common methods of enrichment were gaseous diffusion and gas centrifuge separation. Until recently, gaseous diffusion accounted for just under half of global enrichment capacity (Sovacool, 2011; World Nuclear Association, 2014d). Much of this capacity was originally developed for weapons programs but had been repurposed for commercial uranium enrichment (IAEA, 2000). In the gaseous diffusion process, UF_6 is heated into a gas and forced, under pressure, through a series of porous membranes. The lighter U-235 isotope moves more quickly and has a better chance of passing through the membrane than the heavier U-238 (Sovacool, 2011; World Nuclear Association, 2014e). In this way, a concentration gradient is created. Enriched UF_6 gas is drawn from one end and depleted UF_6 is drawn from the other. This process is repeated in a cascade through as many as 1400 stages before the appropriate level of enrichment is achieved (IAEA, 2000).

All major global gaseous diffusion plants have reached the end of their design lives and have been shut down, with global focus shifting towards more energy-efficient and economical gas centrifuge technologies (World Nuclear Association, 2014d). Gaseous diffusion separation consumes approximately 50 times more energy than gas centrifuge separation to attain the same level of enrichment (World Nuclear Association, 2014d).

In gas centrifuge separation, UF_6 gas is fed into a series of vacuum tubes. A rotor spins rapidly, causing the heavier U-238 isotopes to concentrate toward the outer edge of the cylinder, with a corresponding increase of U-235 near the center (World Nuclear Association, 2014d). The depleted stream is drawn out from the bottom of the centrifuge and the enriched stream is drawn from the top (USEC, 2013). Multiple centrifuges are used in series to attain the required enrichment levels (USEC, 2013).

Although gas diffusion and centrifuge separation have been the most common means of enrichment, other methods may become more economical in the future. One potentially promising technology is Global Laser Enrichment (GLE, formerly SILEX) (World Nuclear Association, 2014d). In this process, UF_6 gas is "exposed to a laser beam that preferentially excites the 235-UF_6 isotope, which enables separation of natural uranium into enriched and depleted uranium" (GE, 2014). The technology may be able to perform enrichment with considerably higher efficiencies than earlier

technologies at a fraction of the cost (GE, 2014; World Nuclear Association, 2014d). In September 2012, the United States Nuclear Regulatory Commission had granted permissions for the construction and operation of a commercial-scale GLE facility at Wilmington, North Carolina (NRC, 2012).

2.4.1.8 Fuel Rod Fabrication

The next stage is fuel rod fabrication. Again, several fabrication methods can be used but the most common are the 'wet' and 'dry' methods.

In the dry method, UF_6 is heated to a vapor and fed into a two-stage reaction vessel. The gas mixes with steam to create a solid uranyl fluoride before being reacted with H_2 which removes fluoride and reduces the uranium to pure microcrystalline UO_2 (World Nuclear Association, 2014e).

In the wet method, UF_6 is injected into water to form uranyl fluoride particulate slurry. Ammonia or ammonium carbonate is added to produce ammonium diuranate or ammonium uranyl carbonate respectively. The slurry is filtered, dried, and heated in a reducing atmosphere to form pure UO_2 (World Nuclear Association, 2014e). While this method is more complex than dry method and has more waste, it offers greater flexibility in terms of the UO_2 powder properties (World Nuclear Association, 2014e).

Enriched UO_2 is roll compacted into small grains and formed into small cylinders. The pellets are baked into a hard ceramic form and then ground into the required size. The cylinders are packed into thin tubes called fuel rods which are bundled together in a fuel assembly (Cameco Corporation, 2014b).

Research in nuclear fuel technology continues. Some advanced nuclear fuels use all-metal rather than ceramic fuels. These allow more cooling, and can therefore safely accommodate higher power densities leading to improved economics (World Nuclear Association, 2014e). Other research areas focus on improving the ability of fuels and fuel cladding materials to withstand the extreme environments of the reactor core, thereby improving safety and performance (World Nuclear Association, 2014e).

2.4.1.9 Reactor

In 2013, 434 nuclear reactors were in commercial operation (World Nuclear Association, 2014c). The construction, operation, and decommissioning of these facilities is a major engineering challenge and represents some of the greatest costs associated with the nuclear fuel cycle (World Nuclear Association, 2013).

Construction

The construction of nuclear power plants incurs considerable economic and energy costs. These include site preparation, construction, manufacture, and commissioning. Nuclear plants require large amounts of steel and concrete as well as electro-mechanical systems to provide electricity, cooling, ventilation, information, control, and communication (World Nuclear Association, 2013).

Sovacool (2008) reports that a typical nuclear plant will require approximately 80 km of piping welded 20 000 times, 1400 km of electrical cables, 150 000 tonnes of concrete, 30 000 tonnes of steel, 1200 tonnes of copper, and 186 000 tonnes of other material.

The construction phase takes four years or more, and construction times are trending upward (Schneider et al., 2014). Between 2004 and 2014, 36 reactors were started with an average construction time of 9.4 years, ranging from 3.8 to 36.3 years (Schneider et al., 2014). This long lead time makes nuclear power particularly sensitive to changing interest rates and cost overruns during construction delays (Sovacool, 2011).

Operation

Nuclear reactors create electricity by harnessing thermal energy released from the continuous fission of atoms in the fuel source. The heat is used to produce steam which drives turbines to produce electricity. The steam turbine technology employed at nuclear power plants is the same as that used to produce electricity in fossil-fuel based power plants (World Nuclear Association, 2014c).

Pressurized water reactors (PWR) are the most common plant design. In a PWR, water is held above 300°C under pressure in a primary circuit and steam is generated in a second circuit through a heat exchanger (World Nuclear Association, 2014c). Alternatively, boiling water reactors (BWR) create steam in a primary circuit above the reactor core (World Nuclear Association, 2014c). Both of these are specific types of light water reactors (LWR), so-named for their use of natural fresh water as a reactor coolant and moderator for the nuclear reactions (World Nuclear Association, 2014c).

Within a LWR reactor core, fuel rods are arranged in fuel assemblies. Control rods made of neutron-absorbing material are inserted or withdrawn from the core to control the reaction rate or stop it (World Nuclear Association, 2014c). Every 1-2 years, the reactors need to be shut down for refueling at which time, $^1/_4 - ^1/_3$ of the fuel assemblies are replaced (World Nuclear Association, 2014c).

There is an incredible amount of primary energy embodied in each fuel rod. If all of the uranium in a fuel bundled could be fissioned, the total energy release would be 800,000 megawatt days per tonne of uranium (MWd/t) compared to 0.35 MWd/t of coal (Adamantiades and Kessides, 2009). Because only U-235 is fissionable and due to other technological limitations, actual burn-up is significantly less (~5%), but is consistently improving. In 1990 in the United States, average burn-up in BWRs and PWRs was ~24,000 and ~34,000 MWd/t respectively, improving to ~42,000 and ~46,000 MWd/t respectively by 2005 (IAEA, 2006). Generation III and III+ reactors achieve burn-up rates of 45,000 MWd/t and better (World Nuclear Association, 2014f).

Compared to construction and decommissioning, plant operation costs can be quite low. Most of the fuel costs are incurred during enrichment and, as such, nuclear energy prices are relatively insensitive to the market costs of uranium (World Nuclear Association, 2013). However, there are documented cases in the United States where continued operation of existing nuclear plants has not been deemed cost-competitive with the construction and operation of new fossil-fuel plants. In these cases, the nuclear plants closed before reaching the end of their design life (Schneider et al., 2014).

Most plants in operation were designed for 30-40 years of use and by mid-2014, the unit-weighted average of the world's reactor fleet was 28.5 years (Schneider et al., 2014). In the U.S., 72 of the 100 operating reactors have received license extensions to 60 years with an additional 19 applications under review (Schneider et al., 2014). It bears noting that none of the 32 reactors that have been shut down in the U.S. achieved their 40 year expected lifespan (Schneider et al., 2014). In France, "only 10-year extensions are granted and the safety authorities made it clear that there is no guarantee that all units will pass the 40-year in-depth examinations" (Schneider et al., 2014). Globally, 39 of the 388 operating reactors are operating beyond 40 years (Schneider et al., 2014).

Backend: Processing and Storage of Waste Stream

Once nuclear fuel has been used in a reactor, it is considered high-level radioactive waste and contains an assortment of long-, intermediate- and short-lived radioactive elements and compounds. Initially, spent fuel is stored at the reactor site in large pools of water, typically for 10 year or more (Sovacool, 2008). The fuel is then loaded into large concrete casks "that provide air-cooling, shielding, and physical protection" (Sovacool, 2008). To prevent corrosion, helium may be used in place of air as the casks' internal atmosphere (Sovacool, 2008).

Since spent fuel represents an important, long-term hazard to human and environmental health, safe storage is a major consideration. Long-term storage is proposed to be in deep, stable geological repositories, but finding an adequate facility has been a continuing problem (Adamantiades and Kessides, 2009). Approximately 12,000 tonnes of high-level waste are generated every year around the world. While this is a small amount relative to the waste streams of other energy technologies (e.g., 2,023,000,000 tonnes of CO_2 released from fossil fuel-based electricity generation in the United States in 2012 alone (EPA, 2014)), the careful handling required makes it especially challenging (Adamantiades and Kessides, 2009).

Reprocessing of spent fuel can partly address the problem of waste storage. To date, almost 90,000 tonnes of used fuel from commercial reactors has been reprocessed and global reprocessing capacity is now at 4000 tonnes/year (World Nuclear Association, 2014g). Reprocessing cannot eliminate the need for long-term storage, but it can simplify the process, both by reducing the volume of high-level waste needing storage and by reducing the overall level of radioactivity in the fuel waste (World Nuclear Association, 2014g).

Decommissioning

World Nuclear Association (2014h) describes three internationally adopted approaches to decommissioning nuclear facilities:

- **Immediate Dismantling**: Dismantling or decontamination activities begin soon after facility closure. Site is then available for re-use once released from regulatory control.
- **Safe Enclosure**: Facility is placed into a safe storage configuration until residual radioactivity has decayed, after which dismantling and decontamination activities occur. This option postpones the final removal of controls for a longer period, usually in the order of 40 to 60 years.
- **Entombment**: Facility is placed into a safe storage configuration that will allow the remaining on-site radioactive material to remain on-site without ever removing it totally. This option may include encasing the facility in a long-lived structure (e.g., concrete) that will last for a period of time to ensure the remaining radioactivity is no longer of concern.

"As of December 2013, 149 power reactors worldwide had been permanently shut down. In total, 16 power reactors have now been fully dismantled; a further 52 are in the process of being dismantled; 59 are being kept in a safe enclosure mode or are awaiting commencement of the final dismantling; three are entombed; and 17 do not yet have a specified decommissioning strategy" (IAEA, 2014).

2.4.2 Pressurized Heavy Water Reactors (CANDU)

Although light water reactors dominate the global commercial reactor fleet, there are a number of alternative, commercially operating designs. The most common alternative designs are the Canadian CANDU reactors. CANDU reactors are pressurized heavy water reactors, so-named for their use of heavy water (deuterium oxide, HDO, ^2H_2O, D_2O) as a reaction moderator in contrast to normal or 'light' water (hydrogen oxide, H_2O) (World Nuclear Association, 2014c).

The key difference between light and heavy water reactors is that the latter is able to use natural uranium (i.e., 0.7% U-235) as a fuel source. In the more common LWR design, criticality cannot be maintained with such a lean fuel. As the reaction proceeds, the water absorbs some of the neutrons released, thereby slowing and eventually stopping the nuclear reaction. Heavy water does not absorb neutrons and the nuclear reactions can be sustained (World Nuclear Association, 2014c). This design feature allows the CANDU nuclear fuel cycle to bypass the costly and energy-intensive uranium enrichment stage required in the LWR nuclear fuel cycle (World Nuclear Association, 2014e).

However, the production of heavy water (99.7% D_2O) from natural light water (0.015% D_2O) is itself an expensive and energy-intensive process required as part of the power plant commissioning (Miller, 2001; World Nuclear Association, 2014c). There is some disagreement in the literature as to whether these high upfront

economic and energetic demands (and corresponding GHG emissions) offset those from uranium enrichment (Andseta et al., 1998; Lenzen et al., 2006; OpenEI, 2012).

CANDU reactors have an additional innovation. Fuel is stored in separate pressurized tubes within the reactor core that can be independently isolated so that fuel bundles can be replaced. This 'hot refueling' feature allows the reactor to be refueled on-line without a reactor shutdown, i.e., while electricity is still being produced (World Nuclear Association, 2014c).

2.4.3 Generation IV Reactors

Most of the global commercial reactor fleet uses Generation II reactor technology. Generation III and III+ advanced reactors are in various stage of development, with the first advanced reactors now operating in Japan (World Nuclear Association, 2014c). Most Generation II, III, and III+ reactors operate in similar ways, using natural or enriched uranium as a fuel source and water as a moderator. The primary innovation in advanced designs is enhanced safety (World Nuclear Association, 2014c).

Work on next-generation reactors has been progressing in recent years, although the world is unlikely to see a Generation IV reactor in commercial operation before 2030 (World Nuclear Association, 2014c; Adamantiades and Kessides, 2009; Sovacool, 2011; GIF, 2014).

Generation IV reactor technologies are being developed with 5 key criteria considered (Adamantiades and Kessides, 2009; Sovacool, 2011):

1. Sustainable energy
    a. Extended fuel availability
    b. Positive environmental impact
2. Competitive energy
    a. Low costs
    b. Short construction times
3. Safe and reliable energy
    a. Inherent/passive safety features
    b. Public confidence in nuclear energy safety
4. Proliferation resistance
5. Physical protection

One of the key innovations is the ability to extract considerably more energy from the nuclear fuel source than older reactors. As mentioned previously, the theoretical upper end of energy available from nuclear fuel is 800,000 MW-days/tonne if all of the uranium in the fuel assembly could be consumed, but actual burn-up is less than 45,000 MW-days/tonne in most commercial light water reactors (Adamantiades and Kessides, 2009). Some Gen-IV reactors are expected to approach 200,000 MWd/t (GIF, 2014). They can achieve this by breeding fissile fuel from previously fertile

material within the reactor, by consuming more of the fissile material in closed fuel cycles, and by realizing higher thermal efficiencies (GIF, 2014).

The volume of high-level waste generated per unit of electricity from Gen IV reactors is considerably lower than in older reactors, and the high-level waste that is generated has a much shorter contaminating life span (World Nuclear Association, 2014c; GIF, 2014).

2.4.4 Small Modular Reactors

In response to the high capital cost and complexity of large power reactors, there has been a push to develop smaller units, typified by the small modular reactor (SMR) (World Nuclear Association, 2014h). SMRs are usually rated at less than 300 MW_e and achieve economies of scale by the number of units produced as opposed to the size of an individual reactor (World Nuclear Association, 2014h).

SMRs benefit from simpler design, may have reduced siting costs, and are provisioned with a high level of passive safety features (World Nuclear Association, 2014h). Smaller reactors may also be suitable for a broader range of applications than large reactors, including powering remote electrical grids where electrical loads are relatively small and variable (World Nuclear Association, 2014h).

At present, three main SMR technologies are being developed: light water reactors, fast neutron reactors and graphite-moderated high temperature reactors (World Nuclear Association, 2014h). LWRs have the lowest technological risk since they are similar in design to most modern reactors. However, fast neutron reactors can be smaller, simpler, and operate longer before refueling (World Nuclear Association, 2014h).

Globally, there are two SMRs in operation, four under construction, and eleven in an advanced stage of development (World Nuclear Association, 2014h).

2.5 GHG Emissions as Reported in the Literature

Since the 1970s, hundreds of nuclear power life cycle assessments have been performed with widely diverging results (Sovacool, 2008). Some differences are due to the particular approach taken in each study (e.g. methodology, completeness, and assumptions) while other differences are due to scenario-specific factors (e.g. primary energy mix, technologies involved, and ore grade). Reports vary in quality, transparency, and uniqueness.

To understand and reconcile these differences, several authors have performed critical reviews, meta-analyses, and harmonization of previous work. This work is presented in Fthenakis and Kim (2007); Sovacool (2008); Beerten et al. (2009); and Warner and Heath (2012) and is discussed in the following sections.

Even after extensive critical review and harmonization, considerable irreconcilable differences remain. Results from Fthenakis and Kim (2007); Sovacool (2008); Beerten et al. (2009) are summarized in Table 2-3. Results from Warner and Heath (2012) are presented in Section 2.5.1.5.

Table 2-3. Life Cycle GHG Emission Intensities as Reported in Three Critical Review Studies. Numbers are presented in units of g CO_2e/kWh

	Ore Grade	Frontend							Power Plant			Waste Management				Total
Study	**% U_3O_8**	**Mining**	**Milling**	**Refining**	**Conversion**	**Enrichment**	**Fuel Fabrication**	**Transport**	**Construction**	**Operation**	**Decom-missioning**	**Reuse or Recycling**	**Temporary Storage**	**Final Disposal**	**Other**	
Results from (Fthenakis and Kim, 2007)																
US Worst	0.05%	5.5		N/A	1.5	20	0.7	N/A	5.6	10.8	5.6		4.8		0.50	**55.0**
US Baseline	0.20%	1.7		N/A	0.9	13	<0.1	N/A	0.58	3.90	0.58		3.3		0.04	**24.0**
US Best	12.7%	0.1		N/A	<0.1	11	<0.1	N/A	0.26	2.50	0.26		1.7		0.18	**16.0**
Studies in (Sovacool, 2008)																
(Andseta et al., 1998)	1.77%	0.43		N/A	0.14	not required	0.11	0.01	2.22	11.90	0.61		N/A			**15.4**
(Barnaby and Kemp, 2007)	0.15%				56.00			N/A	11.50	N/A	35.50		N/A			**103.0**
(Dones et al., 2005)**- BWR**		0.4	1.1	N/A	1.2	7.75	0.08	N/A	N/A	1.04	N/A	0.15	0.08	0.20		**12.0**
(Dones et al., 2005)**- PWR**		0.4	1.1	N/A	1.2	0.4	0.08	N/A	N/A	1.20	N/A	0.20	0.08	0.20	0.14	**5.0**
(Dones et al., 2003a; Dones et al., 2003b)					9.0			N/A	1.15	N/A	N/A		0.8			**11.0**
(Dones et al., 2004)					42.4			N/A	1.2	N/A	N/A		0.9			**44.5**
(ExternE, 1998)					N/A			N/A	11.5	N/A	N/A		N/A			**11.5**
(Fritsche and Lim, 2006)					20.0			N/A	11	N/A	N/A		33			**64.0**
(Fthenakis and Kim, 2007)					shown above			N/A	shown above							
(Hondo, 2005)		1.1		N/A	0.2	15.0	0.7	0.0	3	3.2	0.4		0.7	0.1		**24.2**
(IEA, 2002)					4.86			N/A	2.55	N/A	0.17		4.86			**12.4**
(ISA, 2006)	0.15%	1.80	1.80	N/A	3.70	13.40	1.75	0.03	5.2	13.30	0.50		6.6	2.0		**50.1**
(Rashad and Hammad, 2000)					23.5			N/A	2.0	0.40	N/A		0.5			**26.4**
(Storm van Leeuwen and Smith, 2005)					36.0			N/A	23.5	N/A	34.50		17.0			**111.0**
(Storm van Leeuwen, 2006)					39.0			N/A	24.5	N/A	36.00		17.0			**116.5**
(Storm van Leeuwen and Smith, 2007)	0.06%	24.73		4.96	N/A	5.43	1.45	N/A	20.0	24.4	44.3		28.13			**153.4**
(Tokimatsu et al., 2006)			0.05		0.1	5.1	0.3	0.06	1.30	2.0	0.1		0.99			**10.0**
(White and Kulcinski, 2000)		0.4			8.9			0.2	1.90	2.2	0.01		1.40			**15.0**
Studies in (Beerten et al., 2009)																
(Torfs et al., 1998; Voorspools et al., 2000)	0.20%	0.83		N/A	1.05	1.77	0.05	0.02	2.12	0.80	0.11		N/A		0.97	**7.7**
(Storm van Leeuwen and Smith, 2005)	0.15%	9.25		N/A	2.57	2.94	0.63	N/A	23.78	19.04	35.67		13.75	9.59		**117.2**
(Lenzen et al., 2006)	0.15%	3.77		N/A	3.92	16.1	1.82	0.03	5.38	13.8	1.88		4.01	6.99		**57.7**

N/A = not available
value includes multiple processes
BWR is Boiling Water Reactor
PWR is Pressurized Water Reactor

2.5.1 Critical Reviews and Meta-Analyses

2.5.1.1 Fthenakis and Kim, 2007

Fthenakis and Kim (2007) consider 7 nuclear life cycle analyses performed by 6 authors and organizations representing Australia, Sweden, Switzerland, Japan and the United States. Based on these data, the authors have created estimates for a baseline, best, and worst case life cycle GHG emissions intensity scenario in the US nuclear fuel cycle. These are broken down into the following phases:

- Mining-milling;
- Conversion;
- Uranium enrichment;
- Fuel rod fabrication;
- Construction and decommissioning of the power plant;
- Operation of the power plant;
- Low-level radioactive waste disposal;
- High-level radioactive waste disposal; and
- Deconversion (fluoride is extracted from depleted tailings of enrichment process and uranium is deconverted to U_3O_8 for long term storage (U.S.NRC, 2014)).

Three of the seven life cycle analyses specify that they are considering light water reactors (both PWRs and BWRs) (Hondo, 2005; Storm van Leeuwen and Smith, 2005; Vattenfall, 2014). The others do not state the type of reactor technology considered in the review, but it is likely that they, too, are Generation II LWRs as these are the most common operating reactors.

Fthenakis and Kim (2007) report an overall range in life cycle GHG emission intensity for the nuclear fuel cycle of 16-55 g CO_2e/kWh for the constructed scenarios. Results for each phase and overall emission are shown in Table 2-3.

The authors find that the biggest differences between studies occur in enrichment, production, and operation phases (Fthenakis and Kim, 2007). Enrichment, in particular, contributes 2/3 of the emissions in the worst-case scenario. This is, in turn, due to the enrichment technology used and the primary energy mix (Fthenakis and Kim, 2007). The use of diffusion enrichment as opposed to centrifugal enrichment can increase the energy requirements 30-60 times with an associated increase in emissions (Fthenakis and Kim, 2007).

As discussed in Section 2.4.1.7, all major diffusion enrichment facilities have now closed. The authors anticipate this scenario and estimate that, if all enrichment were to come from centrifugal enrichment, the baseline life-cycle GHG emissions from the US nuclear-fuel cycle would fall to ~12 g CO_2e/kWh (Fthenakis and Kim, 2007).

Upstream electricity source plays important role in emissions during this stage. If the energy required for diffusion is generated by low-carbon sources such as hydro and nuclear, emissions are lower when compared to countries that are fossil fuel dependent.

In terms of emissions from the construction of the power plant, much of the variation can be attributed to the methodology employed in the analysis. The use of steel and concrete represent some of the largest emissions in the construction phase. Studies employing EOI methodology reported 10 to 20 times the emissions from the use of these materials compared to studies using PCA (Fthenakis and Kim, 2007). (Cameco Corporation, 2015a)

Across the globe, the average grade of uranium ore varies significantly, from as high as 17.8% at Cigar Lake Operation, Canada (Cameco Corporation, 2015b), to 0.05% in Australia with a global average somewhere between 0.1% and 0.2% (Fthenakis and Kim, 2007). This difference can affect the emissions from mining-milling considerably. Fthenakis and Kim (2007) calculate the range of emissions from mining-milling to be 0.1-5.5 g CO_2e/kWh based on a range of ore grades from 12.7%-0.05% U_3O_8. This is a factor of 55 difference and represents 10% of the life cycle emissions in the worst-case scenario.

2.5.1.2 Sovacool, 2008

A critical review of previous nuclear LCA studies has been performed by Sovacool (2008). The author reviewed 103 life cycle studies and subjected them to a three-tiered screening process. Studies passing the screening were:

1) Recent
    a. Published after 1996
2) Accessible
    a. In the public domain
    b. Free to access
    c. Published in English
3) Methodologically sound
    a. Relied on published data and/or primary sources
    b. Explained methodology
    c. Transparent regarding data sources
    d. Separated emissions based on life cycle

Nineteen studies passed the screening process and are listed in Table 2-3. Each is broken down into the following phases:

- Frontend
- Operation
- Construction
- Operation
- Decommissioning
- Backend

Where possible, the 'frontend' results listed in Table 2-3 are further broken down after reviewing the source material.

In the 19 studies included in Sovacool's 2008 analysis, GHG emission intensities ranged from 1.4-288 g CO_2e/kWh with a mean value of 66 g CO_2e/kWh. These results include the estimates from Fthenakis and Kim (2007). Two of studies considered heavy water reactors while the rest considered light water reactors (Sovacool, 2008).

Variation in the reported values has been attributed to study completeness, assumed ore grade, type of mine used (open pit, underground, co-mining), primary energy mix, enrichment technology, and a host of assumptions about reactor performance characteristics (Sovacool, 2008). Each of these factors is discussed in greater detail in Section 2.5.2.

In a subsequent publication, Sovacool makes the case that nuclear energy is a poor solution to the GHG emission problem, arguing that emissions from the nuclear fuel cycle will inevitably increase as high-grade ore is used up (Sovacool, 2011). He predicts that emissions from nuclear will match that of natural gas plants by 2050 if its share in the global energy mix remains the same (Sovacool, 2011). Mudd (2014) disagrees, stating that, by his analysis, even large increases in emissions intensity from the mining and milling phase are "unlikely to reach a level which could materially affect the carbon intensity of nuclear power". Mudd (2014) is discussed in more detail in Section 2.6.

2.5.1.3 Beerten et al., 2009

Beerten et al. (2009) recognized the discrepancies between LCA studies, but also the shortcomings of the critical reviews that had been done so far. They are particularly critical of Sovacool (2008) arguing that much of the high-emissions studies found in that review were, in fact, traceable to the same input data and created by the same author, namely Storm van Leeuwen. Beerten et al. (2009) concluded the distribution of results is distorted.

The authors argue that the use of different primary energy mixes and wide-ranging assumptions, estimates, and simplifications in each study make a simple averaging of published values an inappropriate method to establish an overall emission coefficient. Such a number is not reflective of individual situations and is, therefore, of little value to policymakers in making their decisions (Beerten et al., 2009).

To address these issues, Beerten et al. (2009) considers 4 PWR-based nuclear scenarios representing high- medium- and low-range estimates for nuclear life cycle GHG-emission intensity, drawing on data from Torfs et al. (1998); Voorspools et al. (2000); Storm van Leeuwen and Smith(2005); and Lenzen et al. (2006).

Each scenario is broken into the following phases:

- Mining-milling;
- Clean up of mine;
- Conversion;
- Enrichment;
- Fuel fabrication;
- Transport;
- Construction;
- Operation, maintenance and refurbishment;
- Temporary storage;
- Final disposal; and
- Decommissioning

which are each explored in some detail. Where appropriate, the thermal and electrical energy consumption of each phase is examined and the emissions are recalculated based on appropriate emission factors (Beerten et al., 2009). This step makes it possible to adapt scenarios to new situations and recalculate results based on different inputs and primary energy mixes.

Additionally, Beerten et al. (2009) examines other important factors in each study including:

- Ore grade;
- Burn up of the fuel in the reactor;
- U-235 fractions in fuel feedstock, product, and depleted tails;
- Enrichment efficiency in terms of separative work units (SWU) required per kg enriched uranium;
- Mix of enrichment technologies employed; and
- Plant efficiency, capacity factor, and lifetime.

These factors are not consistently reported in the literature.

The following sections provide a detailed assessment of each study included in Beerten et al. (2009).

Belgian Study

Each study considered by Beerten et al. (2009) varied in LCA methodology, completeness, and assumptions. The first, by Torfs et al. (1998) and Voorspools et al. (2000) had the lowest emissions intensity estimate, at 7.72 g CO_2e/kWh. The authors predominantly employed process chain analysis (PCA) LCA methodology.

As will be discussed in Section 2.5.2.1, Warner and Heath (2012) have found that PCA correlates with lower emission estimates. This is not to say that the results are less accurate than when other methodologies are used. Indeed, Warner and Heath (2012) find that studies employing PCA show less variation in emission estimates. Refer to Section 2.5.2.1 for a more complete discussion on this topic.

Voorspools et al. (2000) compare estimates of GHG emissions from construction, maintenance, and demolition of a PWR power plant using Economic Input/Output Analysis (EIO) and PCA. They find that the former results in roughly double the emission estimate (~ 4 g CO_2e/kWh for EIO compared to ~2 g CO_2e/kWh for PCA).

Primary energy in this study is from the European grid with one major exception: the enrichment phase, typically the most energy-intensive, is powered exclusively by a French nuclear plant. This greatly reduces the emissions in that phase and overall.

Storm van Leeuwen and Smith 2005 Study

Next, Beerten et al. (2009) examines two scenarios from Storm van Leeuwen and Smith (2005). The first is based on 'soft ore' at an average grade of 0.15%. The second is a hypothetical future scenario set in a time when the world's easily accessible high-grade ore has been depleted and the nuclear industry is supplied by 'hard ore' at an average grade of 0.01%. Both scenarios assume a Generation II LWR.

Results from both of these scenarios have been strongly contested (Sevior, 2006; Dones, 2007; Beerten et al., 2009; World Nuclear Association, 2014i). Storm van Leeuwen and Smith (2005) employ H-IOA and AEI analyses which have, historically, given unreliable results that may overestimate the emissions in each phase (Warner and Heath, 2012).

Emissions from mining-milling, in particular, are much higher in Storm van Leeuwen and Smith (2005) than in the previous study, especially in the 'hard ore' scenario. The emissions in that phase are reported by Torfs et al. (1998), and Voorspools et al. (2000) to be 0.83 g CO_2e/kWh compared to 105.69 g CO_2e/kWh for hard ore in Storm van Leeuwen and Smith (2005).

The high emission estimates from mining-milling may be attributed, in part, to the choice of LCA methodology, but also to the data source selection for mining-milling energy requirements. Storm van Leeuwen and Smith (2005) base these on data from a 1975 study (Rotty et al., 1975) and make no correction for efficiency improvements in the interim, arguing that the increased complexity of mining due to increased safety standards justifies the use of these old values. Additional data for the mining of 'hard ore' comes from literature reviews published in the 70s (ERDA, 1976; Kistemaker, 1976; Rotty et al., 1975) that consider the mining of non-uranium metals. Mine decommissioning emissions modeled by Storm van Leeuwen and Smith (2005) is based on a hypothetical model that does not correspond to methods found in the literature or in common practice (Beerten et al., 2009).

Although the biggest emissions estimate discrepancies occur in the mining-milling phase, other phases in Storm van Leeuwen and Smith (2005) may be subject to large bias. For example, the emissions associated with plant construction are not based on a material inventory, but on the mass of an entire plant and a range of cost data. Energy per unit mass is used in the calculation and found using correction formula developed in old studies (Bullard et al., 1978; Roberts, 1982), based on older data (DOC, 1967). The formula and data were not developed for use with nuclear facilities (Beerten et al., 2009). Beerten et al. (2009) question the appropriateness of this methodolgy and state that the results most likely overstate emissions.

As mentioned previously, much of the studies considered in Sovacool's 2008 wide-sweeping critical review can be traced back to data from Storm van Leeuwen. Given the doubt surrounding Storm van Leeuwen's analysis (Sevior, 2006; Dones, 2007; Beerten et al., 2009; World Nuclear Association, 2014i; Warner and Heath, 2012) and its widespread influence, it is reasonable to submit that the breadth of nuclear power generation life cycle analyses have been skewed.

Australian Study

Finally, Beerten et al. (2009) analyze data from an Australian study performed by Lenzen et al. (2006). This study draws on some of Storm van Leeuwen's 2005 data but combines it with mining data from other Australian non-uranium mines. The resultant emissions associated with mining-milling are similar to the Storm van Leeuwen and Smith (2005) 'soft ore' results.

The Australian case is based on a 100% coal economy. This results in higher emissions even when the estimated energy requirements for phases are similar between studies. This is because the other studies are at least partly based on low-emissions technologies such as hydroelectric and nuclear (Beerten et al., 2009).

Results, Gaps, and Limitations

The results of the Beerten et al. (2009)'s analysis are found in Table 2-3. Some of the shortcomings identified were:

- No disaggregation of mining techniques;
- Mine rehabilitation missing in all but Storm van Leeuwen and Smith (2005),
    - Energy requirements were some of the largest in the study;
- Lack of detail in important phases,
    - Waste processing,
    - Waste storage,
    - Waste disposal,
    - Plant decommissioning; and
- Systematic underestimation or overestimation of results based on LCA methodology.

2.5.1.4 Collection of Results

Data from the three critical reviews previously discussed are shown in Table 2-3. These data are broken down, when possible, into distinct phases. Where required, the source materials were consulted to complete the disaggregation. For the Tokimatsu et al. (2006) study, Sovacool (2008) provides a range estimates based on changing emissions intensities from the beginning of the study period in 1970 to the end of the study period, in 2000. Table 2-3 includes only the more relevant, recent data from the end of the study period.

Results are summarized graphically in Figure 2-7. This figure does not include the duplicate results from Fthenakis and Kim (2007) or Storm van Leeuwen and Smith (2005) presented in Sovacool (2008). Note that Storm van Leeuwen (2005) may still be over-represented in the data set as their results are presented four times in Table 2-3 and three times in Figure 2-7.

The data is represented in summary in Table 2-4. Note that a median value and interquartile range (IQR) are provided rather than a mean and standard deviation. This is consistent with a subsequent analysis by Warner and Heath (2012) who indicate that these measure are less subject to the influence of extreme outliers.

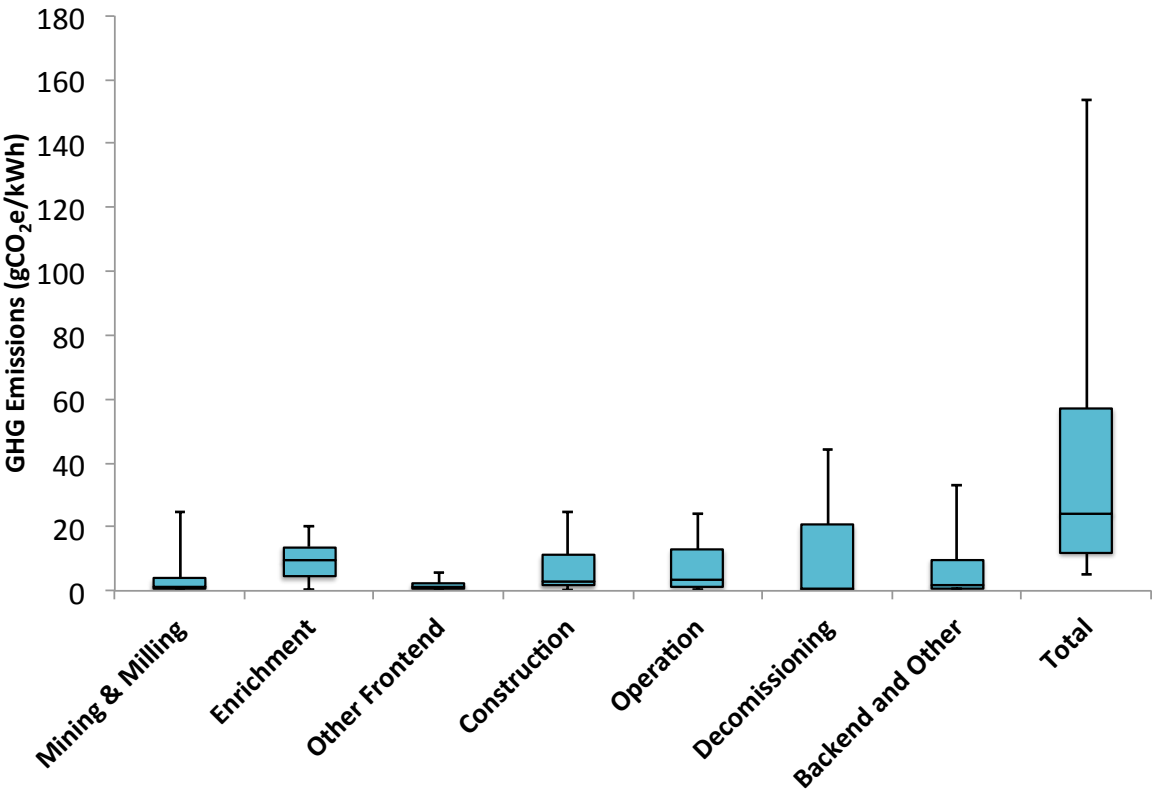

Figure 2-7. Range of GHG Emissions Reported in the Literature from (Fthenakis and Kim, 2007; Sovacool, 2008; Beerten et al., 2009). Boxes represent 2nd and 3rd quartile. Whiskers represent maximum and minimum values.

Table 2-4. Summary Statistics for Life Cycle GHG Emission Intensities as Reported in Three Critical Review Studies, Reported as g CO_2e/kWh

	Mining & Milling	Enrichment	Other Frontend	Construction	Operation	Decommissioning	Backend and Other	Total
Median	1.5	9.4	1.3	2.7	3.6	0.6	1.9	24.1
IQR*	2.9	9.2	1.6	9.4	11.6	20.3	9.0	44.9
Range	24.7	19.6	5.5	24.2	24.0	44.3	32.6	148.4
N**	13	12	12	19	14	14	19	21

*** Interquartile Range is difference between first and third quartiles**
**** Sample size**

2.5.1.5 Harmonization study by Warner and Heath (2012)

In the nuclear LCA studies reviewed above, there is more than a factor of 30 difference between the lowest and highest emission intensity estimates. Additionally, there is a large interquartile range (IQR). In 2012, Warner and Heath (National Renewable Energy Laboratory, Golden CO) published a harmonization study with the aim to determine some of the key causes of this variability and help reduce it with the ultimate goal of helping to inform decision-making.

It is worth noting that, even though emission estimates vary greatly from study to study, the highest estimates for nuclear power are far lower than emission estimates from coal and natural gas power, 220 g CO_2e/kWh vs. 1001 and 477 g CO_2e/kWh respectively (OpenEI, 2012).

Warner and Heath (2012) staged a two-step screening process. Papers passing the first screen were LCAs published since 1980 that evaluated electricity as the product. Sources passing the second, more rigorous screen:

- Evaluated technologies of modern relevance;
- Evaluated uranium mining/milling, conversion, enrichment, and fuel fabrication as part of LCA;
- Reported estimates numerically; and
- Provided enough detail in analyzed system to evaluate data quality.

Studies using EIO analysis or duplicate estimates based on same source material or resubmissions by same author group were excluded (Warner and Heath, 2012).

Of 274 sources considered, Warner and Heath (2012) identified 66 LCA studies that satisfied the screening requirements. Of these, 27 references provided 99 independent estimates of emissions from LWR-based nuclear life cycles. Due to the extensive deployment of LWR reactors globally and the prevalence of data in the literature, these reactors became the focus of the study (Warner and Heath, 2012).

The Warner and Heath (2012) paper was written as part of a larger harmonization effort to resolve major differences in GHG emission estimates for a range of power generation technologies (Warner and Heath, 2012). The authors describe a light harmonization process that includes:

- Adjustment to latest IPCC 100-year global warming potentials
- Proportional adjustment of key plant performance factors including
    - Capacity factor,
    - Operational lifetime, and
    - Thermal efficiency.

Additionally, consistent system boundaries were established by adding missing life cycle phases as required. This boundary is illustrated in Figure 2-8.

The authors note that their study is not a critical review and, as such, has taken the source data at face value without manipulation beyond the harmonization steps outlined above.

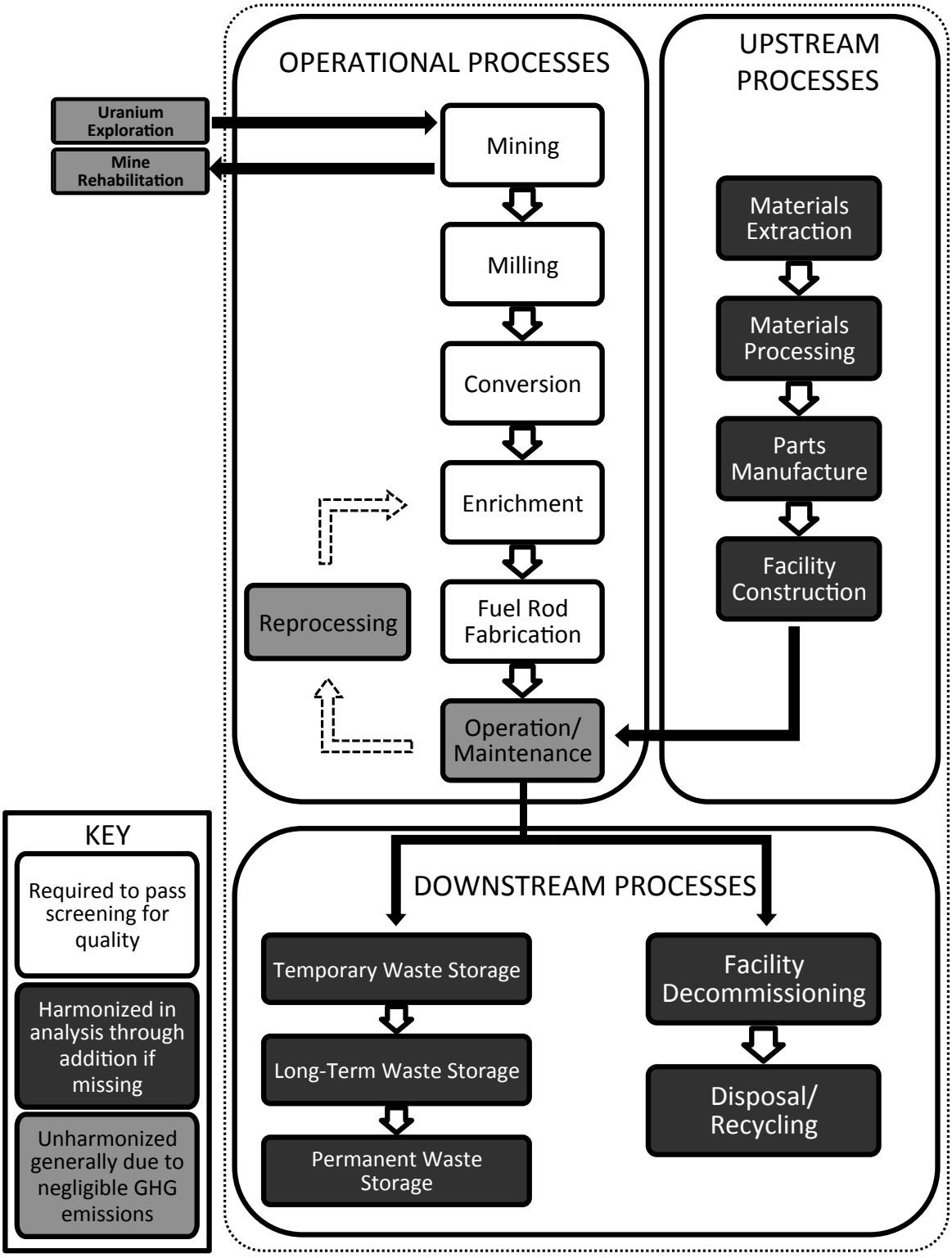

Figure 2-8. Life Cycle Assessment Harmonization System Boundaries, redrawn from Warner and Heath (2012). Items outside of the dotted boundary are not typically considered. This figure is reproduced with permission from the publisher.

Results from the Warner and Heath (2012) study are shown in Figure 2-9 alongside pre-harmonization data and data from the other studies detailed in the previous sections of this thesis. Summary statistics are available in Table 2-5. Results from the harmonization study and other studies considered in this review show similar distributions and many of the primary sources are the same.

Figure 2-9 and Table 2-5 also include pre- and post-harmonization results for nuclear technologies beyond LWR. While not discussed at any length by Warner and Heath (2012), these data are available online at http://en.openei.org/apps/LCA/ as part of the larger harmonization project.

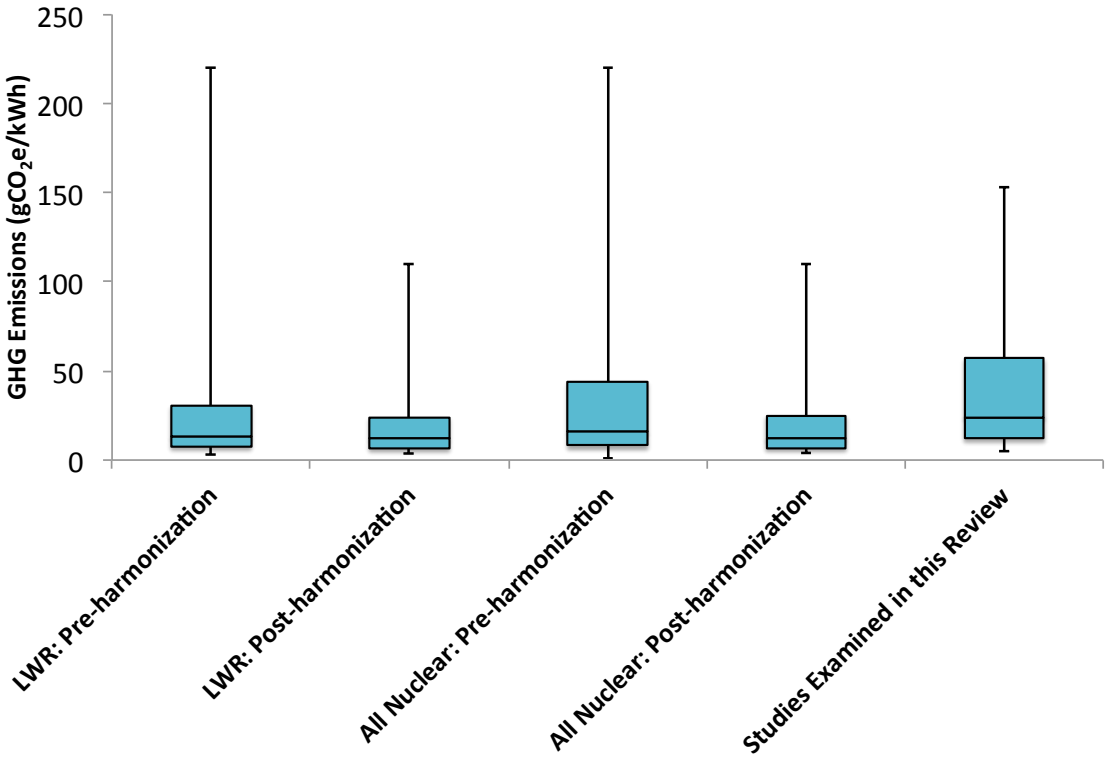

Figure 2-9. Results in Warner and Heath (2012) compared against studies examined in this review (exclusive of results from Warner and Heath (2012)). Boxes represent 2nd and 3rd quartile. Whiskers represent maximum and minimum values.

Table 2-5. Comparison of LCA Results in This Review and Warner and Heath (2012)

	LWR: Pre-harmonization	LWR: Post-harmonization	All Nuclear: Pre-harmonization	All Nuclear: Post-harmonization	Studies Examined in this Review
Median	13	12	16	12	24
IQR	23	17	36	18	45
Range	217	106	219	106	148
N	99	99	130	130	21

The harmonization process resulted in decreased variability in the data set with an IQR reduction of 26% and a total range reduction of 50% for LWR cases (Warner and Heath, 2012). The median, however, changed only slightly, decreasing from 13 to 12 g CO_2e/kWh (Warner and Heath, 2012).

The remaining variability is partly attributable to LCA methodology (i.e. PCA vs. EOI) with much of the rest explainable by physical differences in the following:

- Primary energy mix;
- Uranium enrichment method; and
- Uranium ore grade.

Some effort was made to understand impact of ore grade on emissions, but with less than 50% of LCAs evaluated reporting ore grades, it was not possible to present a robust conclusion about the impact of varying ore grades other than to say that emissions tend to increase as ore grades decrease (Warner and Heath, 2012). The relationship between emissions and ore grade is complicated by the variety of mining methods used and the practice co-mining which makes the mining of low-grade ore possible without a large associated increase in emissions (Warner and Heath, 2012).

Limitations

Warner and Heath (2012) identified gaps in the current literature that remained irreconcilable or missing from their study. These include the following:

- Mine rehabilitation, a potentially significant source of emissions, was not considered in any of the studies passing screens.
- Uranium exploration fell outside of the system boundaries.
- Uranium ore grade was seldom reported and, when it was, did not vary enough between studies enough to develop a robust conclusion.
- Mining methods were often not specified.

Recommendations

Warner and Heath (2012) identified 12 recommendations for future work in LCA research on nuclear power, notably to:

- Follow established minimum guidelines such as ISO 14040 series for conducting LCAs;
- Identify and describe primary source energy mix;
- Look to existing facilities and current practices to refine estimates;
- Include greater detail when describing theoretical life cycle phases;
- Identify mining methods used in ore recovery;
- Refine data on significant life cycle phases; and
- Identify assumed ore grades.

The authors note that, in policy decisions, it is important that impacts beyond GHG emissions be considered, including environmental, social, health, water, and economic impacts.

2.5.2 Discussion of Result Range, Data Gaps, and Methodological Differences

Even after harmonization, GHG emission estimates from nuclear power vary considerably, exhibiting a post-harmonization range of 106 g CO_2e/kWh for all nuclear technologies studied. There are a number of reasons for this variation as outlined in the studies previously discussed. Some of these differences are real, and others are artifacts of the methodology employed.

2.5.2.1 Methodological Differences

The three main LCA methodologies applied to the case studies discussed are process chain analysis (PCA), input/output analysis (IOA), and average energy intensity analysis (AEI). Hybrid analysis employs a mix of PCA and IOA. Results varied widely based on the methodology chosen. Warner and Heath (2012) found that studies employing PCA analysis resulted in a median of 1/3 the value found using hybrid analysis and an inter-quartile range at roughly 1/5 the size.

Warner and Heath (2012) reject studies using the AEI methodology on the basis that this approach has been shown by Beerten et al. (2009), Fthenakis and Kim (2007), and Lenzen et al. (2006) to produce inaccurately high outliers. The use of AEI methodology is reason for some of the extreme variability found in Sovacool (2008), but due to its rejection in Warner and Heath (2012), does not contribute to the range of post-harmonization values found in that study.

While PCA is subject to underestimation due to necessary truncation, EIO is subject to overestimation in nuclear scenarios. EIO analysis is based on cost of materials and, in nuclear applications, the cost of materials rises considerably due to safety requirements. Energy demand increases slightly but does not nearly match the increase in cost (Beerten et al., 2009).

2.5.2.2 Completeness and Inconsistent Boundaries

Some of the variation in Fthenakis and Kim (2007), Sovacool (2008), and Beerten et al. (2009) can be attributed to boundary setting and the completeness of the LCAs being considered. In many cases, important phases such as mine rehabilitation, reactor decommissioning, and waste processing were left out of analyses.

While studies generally had the same gross boundaries, subtle boundaries around individual phases may be inconsistent. In the case of uranium mines, for example, some studies considered only energy used during operation while other considered emissions embodied in the equipment and infrastructure of the facility. Published studies generally did not provide enough information to delineate these subtle boundaries.

The post-harmonization emissions in Warner and Heath (2012) come after reconciliation of LCA completeness and gross boundary adjustment, but stop short of subtle boundary refinement.

2.5.2.3 Scenario-Specific Differences

The explanations for variation have, so far, been artifacts of the approach taken by researchers when conducing LCAs. There are, however, many factors that have a real material impact on GHG emission intensity, some of which are listed in Table 2-6.

Table 2-6. Physical Factors Affecting GHG Emissions in Nuclear Fuel Cycle

Factor	Scope of Impact
Mining Method	Open pit mining has larger footprint and the potential to release more methane than underground and in situ leaching (ISL) (Sovacool, 2008)
Mine Location	Remote mines tend to depend more heavily on diesel generators for electricity, increasing emissions intensity (Sovacool, 2008)
Ore Grade	Higher ore grades are associated with lower emissions in the mining/milling phases and in mine rehabilitation (Sovacool, 2008; Warner and Heath, 2012)
Enrichment Technology	Gaseous diffusion is significantly more energy-intensive than gas-centrifuge (Weisser, 2007)
Primary Source Energy Mix	Fossil-fuel based energy economies emit significantly more GHGs when compared to renewable and nuclear-based energy economies for the same processes (Weisser, 2007)
Reactor Technology	LWRs are most common globally. CANDU reactors do not require uranium enrichment but do have include energy-intensive production of heavy water (Sovacool, 2011) Gen IV reactors are able to extract significantly more energy from the same amount of fuel compared to conventional reactors and have increased capacity to use nuclear waste as an energy source (World Nuclear Association, 2014c). This reduces overall emission intensity.
Process Efficiency	The efficiency of specific facilities along the process chain will have an impact on emissions. This includes milling efficiency enrichment process efficiency and power plant burn up, capacity factor, and thermal efficiency (Beerten et al., 2009).

2.6 Mining-Milling Emission Estimates for McArthur River, Rabbit Lake, McClean Lake

Mudd (2014) has previously estimated emissions intensities for the mines included in this study along with 7 facilities. Using these, Mudd (2014) has generated the following regression equation for carbon intensity of uranium mining versus ore grade:

$$\boldsymbol{GHG\ intensity}\left(\frac{\boldsymbol{kg\ CO_2}}{\boldsymbol{kg\ U_3O_8}}\right) = \mathbf{15.493}\ (\boldsymbol{ore\ grade})^{\mathbf{-0.357}}\ (\boldsymbol{R^2} = \mathbf{0.555}) \qquad \mathbf{(2.5)}$$

where ore grade is expressed as % U_3O_8.

In preparation of his estimates, Mudd (2014) did not consider commissioning and decommissioning, emissions embodied in reagents and infrastructure, transport of workers and material, or emissions from the use of explosives (Mudd and Diesendorf, 2008). Further, it is unclear how Mudd (2014) derived the ore grade utilized in the study.

2.7 Life Cycle GHG Emissions of Other Energy Sources

A full discussion of life cycle processes and emissions of other electrical energy sources are beyond the scope of this literature review. Major aspects of prominent technologies, associated emissions, and implementation challenges are briefly reviewed.

As with nuclear, the reported range of GHG emissions from different power generation technologies range considerably. The Warner and Heath (2012) study discussed previously is part of a broader harmonization project aimed to provide more accurate estimates of emissions from different power generation technologies. To date, GHG emissions from bio-power, solar, wind, and coal power have been harmonized in the same way as nuclear power. These results are available on the web at http://en.openei.org/apps/LCA/ (OpenEI, 2012) and are presented in Figure 2-10.

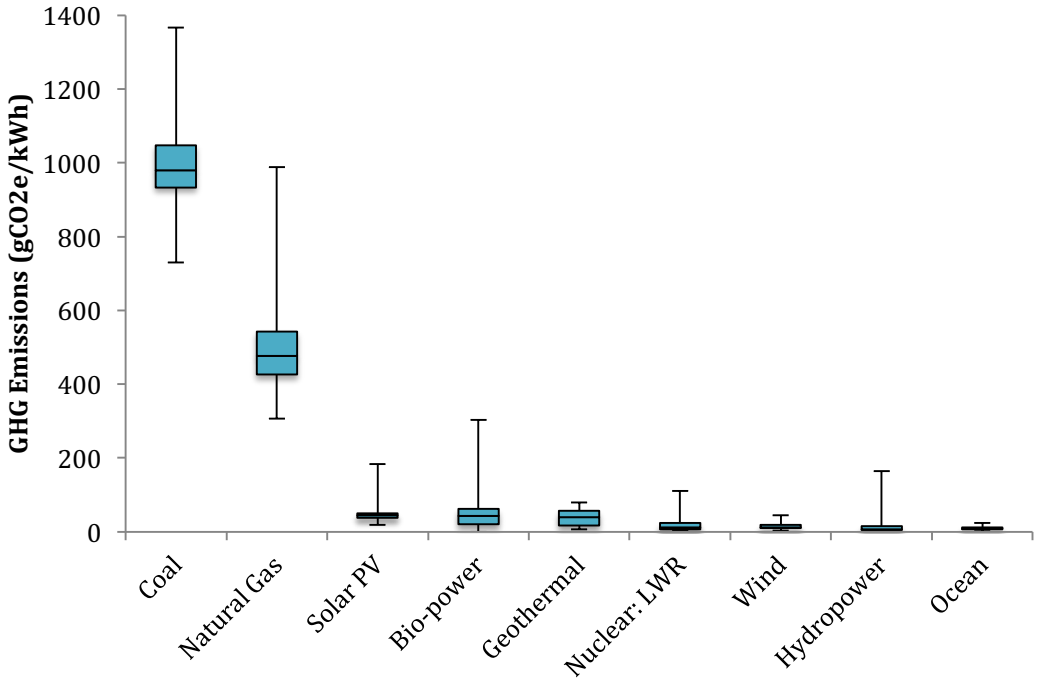

Figure 2-10. Range of Life Cycle GHG Emissions from Various Energy Sources from OpenEI (2012). Boxes represent 2nd and 3rd quartiles. Whiskers represent total range.

To get a better sense of the emissions from non-fossil fuel technologies, Figure 2-10 is reproduced with natural gas and coal excluded in Figure 2-11.

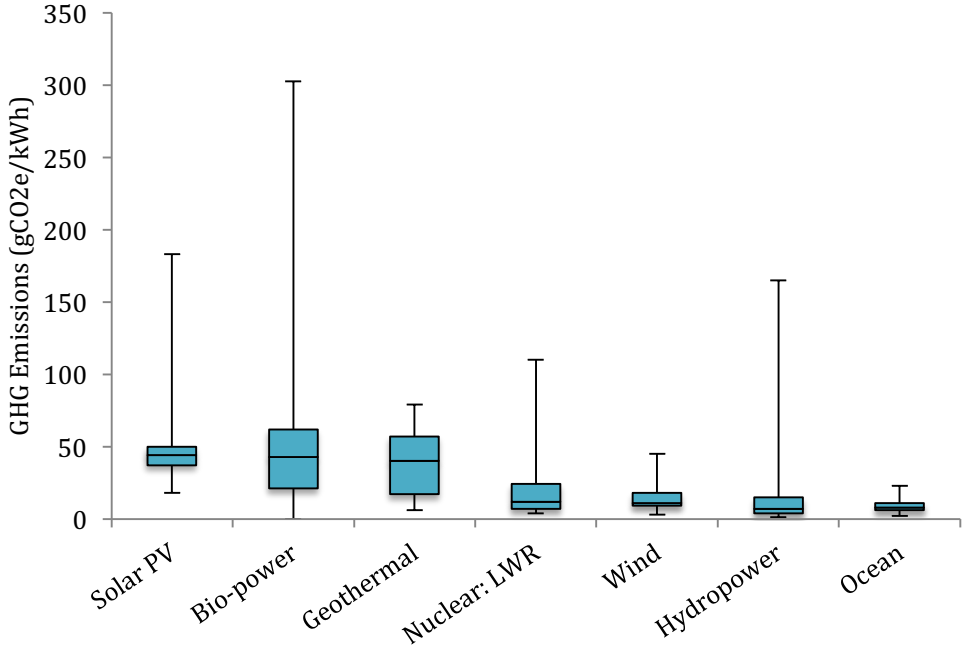

Figure 2-11. Range of Life Cycle GHG Emissions from Various Energy Sources (excluding coal and natural gas) from OpenEI (2012). Boxes represent 2nd and 3rd quartiles. Whiskers represent total range.

Where possible, harmonized results are included in the preceding figures. The ranges reported for geothermal, hydro, ocean, and natural gas are pre-harmonization values.

Although a large range of values are reported for each technology, it is clear that every alternative to fossil fuels is superior from a GHG-emission perspective. Nuclear power appears emissions-competitive with other low-carbon energy sources.

2.7.1 Coal and Natural Gas

Coal and natural gas are the most dominant global energy sources and have the highest GHG emission intensities, up to 100 times the emissions from some renewables (OpenEI, 2012).

In contrast to nuclear, wind, solar, and hydroelectric power, the majority of life cycle GHG emissions occur during operation of the coal and natural gas power plants when the fuel source is burned. For the average coal power plant, some 97.3% of life cycle CO_2 emissions come from power generation, 1.7% from transportation, and 0.9% from mining (Spath et al., 1999). For combined cycle natural gas power plants, 74.6% of CO_2-equivalent emissions come from plant operation and 24.9% from natural gas production and distribution (Spath and Mann, 2000). 11.7% of CO_2-equivalent emissions from natural gas production and distribution come from the release of CH_4 to the atmosphere during these processes (Spath and Mann, 2000).

2.7.2 Wind and Solar

Wind and solar power are often thought of as the most promising low-emission alternatives to fossil fuels, but are faced with several major barriers including large land requirements for utility-scale installations, intermittency issues, high costs, and the inability of old infrastructure to support decentralized and dynamic energy systems.

The intermittency of wind and solar resources may become easier to manage the more they are deployed and interconnected due to geographical smoothing, the balancing of regional power fluctuations over a large area (Sovacool, 2011). Other responses to the intermittency problem include grid-scale battery storage, pumped hydro, compressed air, and molten salt technologies (Sovacool, 2011).

Other challenges are being systematically addressed by government and industry. Costs of solar modules are dropping rapidly and cell efficiencies are continuously increasing with an associated decrease in GHG emission intensity (Fthenakis et al., 2008).

Some major advantages of wind and solar technologies are their modular nature and short lead times. Solar technology can go through several generations of innovation in the time it takes to commissions a nuclear power plant (Sovacool, 2011). As such, it is adaptable to changing power demands and changes in grid infrastructure.

It is worth noting that, with both of these technologies, up to 99% of the life cycle GHG emissions occur before any electricity is generated (Raadal et al., 2011).

2.7.3 Hydropower

Hydropower is, by far, the largest renewable energy source being harnessed for electricity production today and, unlike solar and wind power, can consistently provide large amounts of base load power. Commercial facilities range in capacity from 200 kW to 6.8 GW (Sovacool, 2011).

The large range in reported emissions intensities can be almost entirely attributed to the flooding of land for reservoirs (Raadal et al., 2011). Many large hydroelectric projects necessitate the flooding of land upstream of the facility. The decomposition of biomass in these flooded regions is a significant source of CO_2 and CH_4 emissions. The magnitude of these emissions is related to the total area flooded, the previous land use, and the regional climate (Raadal et al., 2011). Research on GHG emissions from flooded reservoirs is at an early stage (Raadal et al., 2011).

In contrast to large reservoir-based facilities, run-of-the-river hydro requires little-to-no flooding and, therefore, tends to have much lower emissions (Raadal et al., 2011).

2.7.4 Bio-power

Energy can be generated from biomass in a number of ways. These can include the combustion of biogas from landfills or other anaerobic digestion processes, combustion of a biomass feedstock with or without gasification, or co-firing of biomass with another fuel such as coal or natural gas (Sovacool, 2011).

Depending on the fuel source, the GHG emissions from bio-power can range considerably. In landfills, for example, landfill gas can be captured and burned to produce electricity. This converts the highly potent CH_4 to CO_2, effectively avoiding GHG emissions. On the other end of the spectrum, energy crops are sometimes used as a feedstock. These require more energy in growing and processing and may displace food crops (Sovacool, 2011).

In each case, the actual combustion of the fuel source is carbon-neutral since the biomaterial has necessarily removed the equivalent amount of CO_2 from the atmosphere during its growth.

2.7.5 Geothermal and Ocean Power

Geothermal and ocean power tend to be site-specific and have an overall low contribution to global electricity supply. This may be subject to change as research and development in both in these areas continues, but for the time being, they will not be included in any further discussion.

2.7.6 Energy Efficiency and Demand-Side Management

As an alternative to investment in new power sources, energy efficiency measures can be an effective and economical way to reduce power consumption and GHG emissions. Indeed, efficiency may be the low-hanging fruit of energy and emission management. Measures such as lowering thermostats, light bulb replacement, improved HVAC, better vehicle mileage, and implementation of passive heating and cooling systems can reduce emissions while saving money (Sovacool, 2011).

According to Sovacool (2011), investing in efficiency:

- mitigates against uncertainty;
- reduces wear and maintenance on the existing fossil fuel chain;
- depresses costs of locally used fuels such as:
    - uranium, oil, coal, and natural gas;
- reduces demand across peak hours;
- lessens pollution;
- improves reliability of existing generators; and
- requires no intervention by system operators.

Efficiency measures have already proven effective. In the mid-70s, the oil crisis provided incentive for efficiency standards to increase across a number of areas resulting in improved economy and suppressed emissions (Sovacool, 2011). Although there are limits to the application of efficiency measures, there are presently many areas where they can still be effective (Sovacool, 2011).

3. Methods

This thesis presents a detailed study of life cycle greenhouse gas (GHG) emissions during the uranium mining-milling phase of the nuclear life cycle for three paired mining-milling operations in northern Saskatchewan (SK), shown in Figure 3-1. These facilities are operated by Cameco Corporation (Cameco) and AREVA Resources Canada Inc. (AREVA). Each facility is described in Section 3.1.

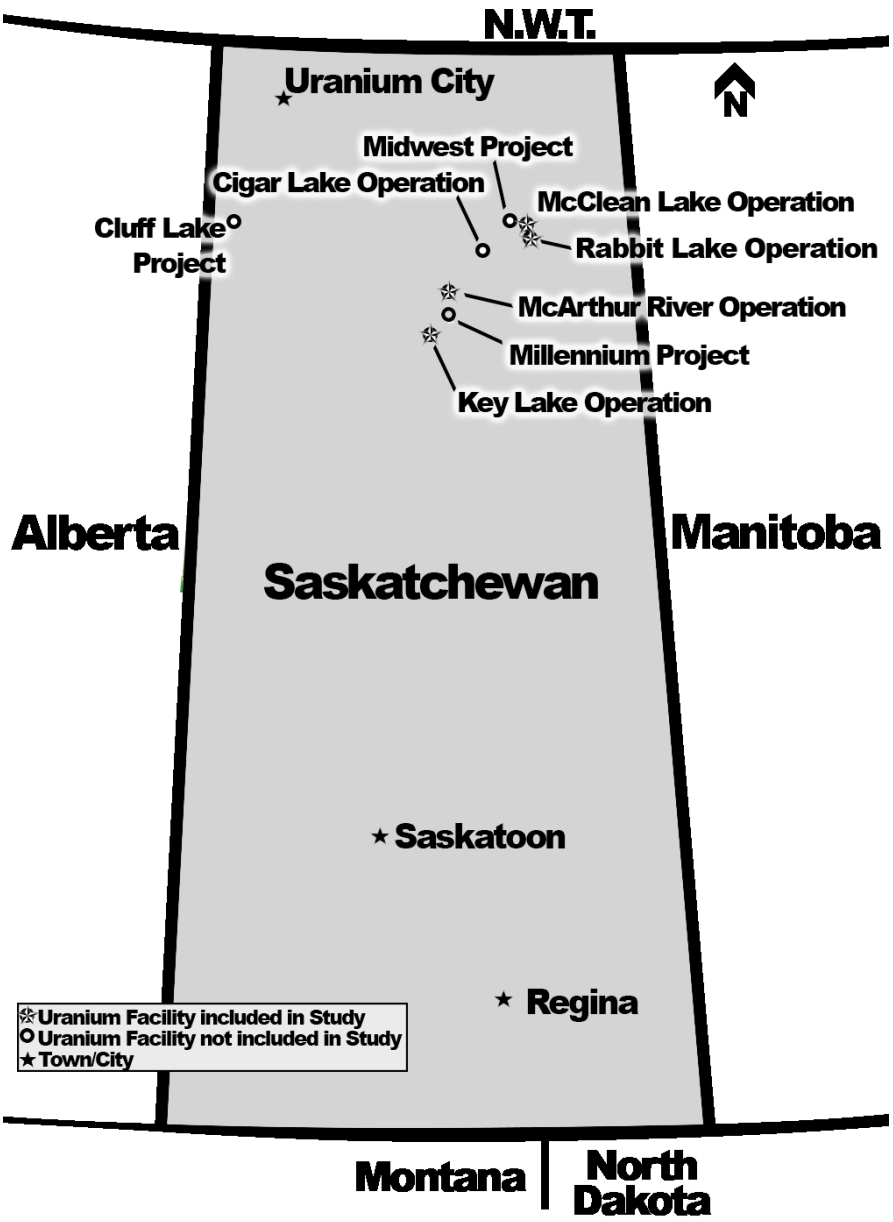

Figure 3-1. Map of Major Uranium Deposits, Mines, and Mills in Saskatchewan

3.1 Facility Profiles

3.1.1 McArthur River Operation-Key Lake Operation (Cameco)

This study includes data from operation at McArthur River-Key Lake from 2006-2013 inclusive. In many cases, data outside of this temporal boundary was available and was also included.

McArthur River Operation (McArthur River) is located approximately 600 km north of Saskatoon by air and 790 km north by truck (Google Maps, 2014). Raisebore mining began in 1999 (Cameco Corporation, 2013b). Ore is processed underground yielding a slurry which is pumped to surface, loaded into specially designed containers, and trucked to the Key Lake mill (Cameco Corporation, 2013b; Cameco Corporation, 2013b).

As of Dec. 31 2013, a total of 114,195,511 kg U_3O_8 had been produced at McArthur River and as of that date, proven and probable reserves of 163,519,142 kg U_3O_8 remain. According to the current life-of-mine plan, ore reserves will be exhausted in 2035 (Cameco Corporation, 2012b; Cameco Corporation, 2013b). Total reserves and mine lifespan are updated annually.

Key Lake Operation (Key Lake) is located approximately 570 km north of Saskatoon by air and 710 km by truck (Google Maps, 2014). It began production in 1983, milling ore from the Gaertner and Deilmann open pits until 1997 (Cameco Corporation, 2013c).

In 2000, the Key Lake mill began to process ore from the McArthur River deposit, 72 km away by truck (Cameco Corporation, 2013c). From 1983-2000, Key Lake shipped 78,492,376 kg U_3O_8 and from 2000-2013, it shipped 77,737,681 kg U_3O_8 sourced mainly from McArthur River (Cameco Corporation, 2012c). Assuming that Key Lake processes the remaining reserves of McArthur River's uranium at an average recovery rate of 98.5%, it will ship another 161,000,000 kg U_3O_8.

3.1.2 Rabbit Lake Operation (Cameco)

The Rabbit Lake facility began operation in 1975 and is: "the longest operating uranium production facility in North America, and the second largest uranium mill in the world" (Cameco Corporation, 2014c). It is located approximately 670 km north of Saskatoon by air and 820 km by truck (Google Maps, 2014).

Rabbit Lake Operation (Rabbit Lake) mill has previously obtained ore from Rabbit Lake open pit, Collins Bay A- B- and D- zones, and currently receives ore from Eagle Point underground mine using a drill and blast method (Cameco Corporation, 2013c). From 1975-2013, Rabbit Lake produced 86,227,910 kg U_3O_8 and additional production of 8,800,000 kg U_3O_8 is expected from 2014-2018 (Cameco Corporation, 2013a; Cameco Corporation, 2013d; Cameco Corporation, 2015a). This study includes data from operation at Rabbit Lake from 2006-2013 inclusive.

3.1.3 McClean Lake Operation (AREVA)

McClean Lake Operation (McClean Lake) began in 1995 with the mining of JEB open pit and the construction of the JEB Mill. AREVA claims that McClean Lake is the most technologically advanced uranium mill in the world, capable of processing ore from grades of less than 1% to 30% without dilution (AREVA Resources Canada Inc., 2014b). It is located approximately 700 km north of Saskatoon by air and 830 km by truck (Google Maps, 2014).

From 1995-2008, five open pits were mined out and the mill processed their ore from 1999-2010. During this time, approximately 22,700,000 kg U_3O_8 was produced (AREVA Resources Canada Inc., 1995-2010).

In 2005, the mill began an expansion project to increase its production capacity and allow it to receive ore from Cigar Lake underground mine, the world's largest undeveloped high-grade uranium mine. The McClean Lake mill is expected to process all of Cigar Lake's 98,300,000+ kg U_3O_8 (Cameco Corporation, 2012a; AREVA Resources Canada Inc., 1995-2010).

In July 2010, the mill shut down as uranium stockpiles were depleted and expansion activities continued (AREVA Resources Canada Inc., 1995-2010). This study includes data from operation at McClean Lake from the beginning of construction until the depletion of ore stockpiled from onsite open pits, 1995-2010 inclusive.

3.1.4 Facilities Excluded from LCA

In addition to the above-mentioned facilities, AREVA has provided details for the decommissioning of Cluff Lake Project (Cluff Lake), an operation that included underground and open pit uranium mines as well as a uranium mill. Mining and milling at Cluff Lake ended in 2002 and decommissioning activities began soon after. In 2013, the last buildings were demolished and site occupancy ceased. Active decommissioning is now complete and the site is in a period of long-term monitoring (AREVA Resources Canada Inc., 2014c). Cluff Lake Project is not considered for detailed analysis for lack of operational data, but the recent decommissioning activities are used to help validate estimates of GHG emissions for decommissioning for the facilities considered as part of this study.

Cigar Lake Project (Cigar Lake) is a high-grade underground uranium mine located 69 km south of the McClean Lake mill, where ore slurry from the project will be shipped and processed (Cameco Corporation, 2012a). The mine began construction in 2005 and its first shipment of ore slurry arrived at McClean Lake mill in March 2014 (Cameco Corporation, 2012a; Cameco Corporation, 2014e). Cigar Lake is expected to reach full-scale production in 2018 (Cameco Corporation, 2014e). Like Cluff Lake, this facility is not considered for detailed analysis for lack of operational data.

Two other major uranium projects, the Midwest Project and the Millennium Mine Project, are under development in SK but are not yet under construction. They too are excluded from analysis.

Historical uranium projects in SK include Gunnar Mine (1955-1963), Lorado Mill (1957-1960), and Eldorado Mine (1953-1982) (SRC, 2014; McBain, 2007). These are excluded from analysis for lack of data and because they do not reflect the current state of the industry, which this study aims to capture.

3.2 Data Collection

The mine-mill operators, Cameco and AREVA, have collaborated in this research by providing multiple years of data related to emissions-relevant activities including energy consumption, reagent consumption, transportation, mining and milling processes, facility history, infrastructure, and more. Additional data was obtained from the ecoinvent life cycle database (ecoinvent Centre, 2013).

It is assumed that all data provided by the mine-mill operators is accurate and has been validated by each company's internal auditing program. Unless otherwise stated, data from the operators is not explicitly validated as part of this study.

Where data was not available, assumptions were made to fill the gaps. The associated uncertainty is assessed according to the methodology discussed in Section 3.5. The following sections note specific instances where the methodology employed deviates from that described in Section 3.5.

The life cycle analysis requires the consideration of construction, operation, exploration, and decommissioning activities as well as emissions embodied in infrastructure, equipment, and materials.

3.2.1 Construction

Construction activities include the transport of construction materials and employees to construct buildings, earthworks, and roads. At uranium mining facilities, it also includes development of the open pit or underground mine.

Since most construction activities occurred before the operating companies had begun systematically assessing and reporting their energy consumption and GHG emissions, there are no accessible records for early construction activities. In the absence of this data, emissions are estimated by comparison with similar facilities for which data or qualified estimates are available.

Energy consumption data during construction of McClean Lake Operation is partially available. Construction activities at McClean Lake include development of both the open pit mines and the uranium mill. Similar activities were undertaken in the development of both Rabbit Lake and Key Lake mills, both of which originally had open pit mines (Cameco Corporation, 2001-2013). Data from McClean Lake is used

to inform emission estimates for direct and indirect energy consumption at these facilities.

Cigar Lake is an underground uranium mine with some similarities to McArthur River including a freezing program, the use of mining techniques that isolate employees from contact with ore, and an ore crushing and grinding circuit located in the underground facility (Cameco Corporation, 2012a). Some site-specific factors complicate the comparison of the two facilities, most notably setbacks at Cigar Lake resulting from three water inflow incidents between 2006 and 2008 (Cameco Corporation, 2012a). Data for energy consumption and employee transport for Cigar Lake are available for nine years (Cameco Corporation, 2014d). The construction estimate for McArthur River is informed by emission from Cigar Lake during this period, but also takes into account the differences between the projects.

The Millennium Project is a proposed underground mine located approximately 600 km north of Saskatoon, midway between Key Lake and McArthur River (Cameco Corporation, 2013e). GHG emission estimates for each production phase are provided in the project Environmental Impact Statement for two scenarios: 1) electricity provided to site in year two through decommissioning; and 2) electricity is not provided to site through life of mine (Cameco Corporation, 2013e). Cameco states that the emission estimates are highly conservative (Cameco Corporation, 2013e). These data are not used directly to estimate emissions for any of the facilities included in this study, as it is not directly comparable to any of those included. Rather, the data is used to validate the estimates that are used in the current study.

The methods used to calculate emissions from direct and indirect energy consumption during the construction phase at these facilities introduces uncertainty to the calculated result. This uncertainty is included and calculated as described in Section 3.5.

3.2.2 Infrastructure and Equipment

It is not possible to directly assess all of the materials present in each building and piece of equipment. However, a rigorous LCA using PCA requires these quantities to be estimated.

Estimates for materials used in the structure of buildings are based on the area and volume occupied by buildings at each site. For Cameco facilities, this information is available in the Preliminary Decommissioning Plans. For AREVA facilities, building area is estimated from site layout drawings found in annual regulatory reports. Building heights for AREVA facilities are estimated based on similar buildings at Cameco sites and on observations made during a site tour. The mass of steel, aluminum, etc. used in the building structure per building unit volume are taken from ecoinvent Centre (2013).

Stationary equipment consists mostly of tanks, pipes, pumps, and hoists. Material types are listed in construction drawings, and mass in tanks and pumps are estimated

based on their geometry as presented in the drawings. Tanks are modeled as cylinders, cones, and/or right rectangular prisms. Tanks are assigned a nominal wall thickness of 6.4mm. Where tanks are rubber lined, the lining is assigned a nominal thickness of 3.2mm. Pumps are modeled as solid steel cylinders. Material estimates based on these are then increased by 40% to account for other equipment such as cranes, ladders, stairs, piping, etc. The material estimate is further increased by 10% to account for equipment replacement.

For facilities where construction drawing were not available, materials usage is estimated. Based on the amount of materials used in the Key Lake mill, a *material intensity* is developed:

$$\boldsymbol{Material\ Intensity}\left(\frac{\boldsymbol{kg\ material}}{\boldsymbol{m^3 building}}\right) = \frac{\boldsymbol{Total\ Materials\ Estimated\ from\ Drawings\ (kg)}}{\boldsymbol{Total\ Volume\ of\ Facility\ (m^3)}} \qquad \textbf{(3.2)}$$

Material estimates for buildings without drawings are then generated based on applying this *material intensity* to the building's known volume:

$$\boldsymbol{Material\ Estimate} = \boldsymbol{Material\ Intensity} \times \boldsymbol{Building\ Volume} \times \boldsymbol{Intensity\ Factor} \qquad \textbf{(3.3)}$$

The *intensity factor* varies between 0 and 1 depending on the relative amount of equipment in a building compared to the Key Lake mill buildings. An *intensity factor* of 1 indicates that the building has a similar amount of equipment in it as a mill building whereas an *intensity factor* of 0 describes an empty building. The choice of this factor is based on the activities undertaken in these buildings and also on information gathered during site tours.

In addition to the equipment and materials discussed above, the items listed in Table 3-1 are also included for analysis.

Table 3-1. Additional Materials and Equipment Included in Analysis

	Data Source	
	McArthur River, Key Lake, Rabbit Lake	McClean Lake
Materials		
Concrete in Foundations	(1)	*
Polyethylene Piping around Site	(1)	*
Fuel-Burning Equipment		
Small Boilers, Vaporizers, and Heating Units	(2)	(2)
Large Boilers, Vaporizers, and Heating Units	(2)	(2)
Diesel Generators	(2)	(2)
Vehicles		
Light Construction/Mining Equipment	(3)	(2)
Heavy Construction/Mining Equipment	(3)	(2)
Small Mobile Equipment	(3)	(2)
Buses	(3)	(2)

(1) Facility-Specific Preliminary Decommissioning Plans (Cameco Corporation, 2013h; Cameco Corporation, 2013i; Cameco Corporation, 2013j)
(2) Facility-Specific Annual Reports (AREVA Resources Canada Inc., 1995-2010; Cameco Corporation, 2001-2013)
(3) Proprietary Data Provided by Facility Operator
* Data unavailable. Estimate based on materials at Key Lake/total building footprint

3.2.3 Operational Activities

Table 3-2 summarizes the operational data requested and received for each facility. Data for McArthur River, Key Lake, and Rabbit Lake was most often available from 2006-2013 with some additional data available for earlier years. Data for McClean Lake was available from 1995-2010. For all facilities, data gaps are generally more common in early years.

Table 3-2. Operational Data Requested and Received

Data Requested	Data Source	Years of Data Available			
		McArthur River	Key Lake	Rabbit Lake	McClean Lake
Operational Parameters					
People on Site (man-days worked)	(1,2)	8	9	5	15
Production Data					
Tonnes Ore Produced/Processed	(1)	14	13	8	16
kg U_3O_8 Equivalents Processed/Shipped	(1)	14	13	8	16
Energy Consumption Data					
Electricity	(2)	8	9	8	11
Diesel	(2)	8	9	8	8
Propane	(2)	8	9	8	12
Gasoline	(2)	8	9	8	12
Transportation Data					
Flight Schedules	(2)	8	7	7	0*
Freight Reports	(2)	8	7	7	12
Fugitive Emission Data					
Domestic Wastewater Generation	(1)	0**	0**	0**	0**
Liquid/Solid Waste Generation	(1,2)	9	9	8	13
Process Emissions	(3)				
Concrete Usage (within mine)	(1)	12	N/A	8	N/A
Reagent Consumption	(1)	4	10	8	15
Explosive Usage	(2)	7	N/A	7	16

(1) Facility-Specific Annual Reports (AREVA Resources Canada Inc., 1995-2010; Cameco Corporation, 2001-2013)
(2) Proprietary Data Provided by Facility Operator
(3) Calculation - Direct CO_2 emissions from reaction of carbonates with sulfuric acid added during milling process – carbonates present in ore and reagents - assumes 100% reaction
N/A – Not Applicable
* None available, amount estimated based on number of people on site compared to McArthur River, corrected for relative distance from Saskatoon
** Estimated assuming 255 L/day generation per man-day worked

3.2.4 Corporate Activities

For both operators, corporate headquarters (HQ) are located in Saskatoon, SK. Emissions from fuel and electricity consumption at HQ are allocated to each mine and mill.

Proprietary data for natural gas and electricity consumption at corporate headquarters was supplied from 2006-2013 by Cameco and from 2003-2013 for AREVA.

Emissions from Cameco's headquarters are allocated one quarter each to Cameco's four main SK operations: McArthur River, Key Lake, Rabbit Lake, and Cigar Lake over the time period where data is available. Emissions from AREVA's headquarters are allocated one half each to Cluff Lake and to McClean Lake.

No other emissions-relevant activities are included as they are expected to fail the cut-off criteria established in Section 1.3.3. As an example, materials used in the construction of corporate buildings are excluded from analysis. The footprint of corporate buildings is very small compared to the building footprint at the mines and mills. The latter contributes less than 0.2% to the emissions total.

3.2.5 Exploration

Exploration activities are typically excluded from life cycle assessments for several reasons: 1) their overall impact is expected to be very small; 2) it is difficult to allocate exploration activities to specific facilities as these activities are ongoing at developed and undeveloped sites, many of which will never be developed; and 3) it can be difficult to collect data from exploration activities for the purposes of life cycle analysis (Storm van Leeuwen and Smith, 2007; Durucan et al., 2006; Awauah-Offei and Adekpedjou, 2011).

However, to ensure completeness, exploration activities are included in this study. There is relatively little data available regarding exploration and so a pseudo-economic input/output analysis is used rather than process chain analysis.

Data on uranium exploration spending by Cameco from 1990 onward is accessible. These costs are allocated one third each to McArthur River, Eagle Point, and Cigar Lake. This is considered a conservative estimate due to ongoing exploration activities being undertaken by the company at other locations. It is assumed that exploration activities are similar to decommissioning activities in that emissions for both are driven by 1) energy consumption in diesel-burning equipment; and 2) employee transport. Under this assumption, every dollar allocated to exploration is assigned the same emissions as a dollar spent on decommissioning at the same facility.

The uncertainty introduced to the system model because of this approach is relatively large compared to the overall GHG emissions from exploration activities. There is a factor of ~5 difference between the low and high bounds of the estimation when the

uncertainty methodology discussed in Section 3.5 is applied. As will be shown in Section 4.4, the effect on the full life cycle result is so small as to obviate the need for a more precise estimate.

No AREVA data was used to estimate exploration emissions for McClean Lake. Instead, it is assumed to be similar to the other facilities.

3.2.6 Decommissioning

Decommissioning activities are described in each site's Preliminary Decommissioning Plan (PDP), a document submitted to the Saskatchewan Ministry of Environment. The PDP is updated in maximum 5-year intervals. This document, and the associated Preliminary Decommissioning Cost Estimate (PDCE), contain estimates for heavy equipment use, energy consumption, major material requirements (e.g. lime, concrete), and employee transportation during decommissioning activities.

PDP and PDCE documents are provided by Cameco for McArthur River, Key Lake, and Rabbit Lake. AREVA has provided a decommissioning plan summary for McClean Lake and a detailed decommissioning report for Cluff Lake. Cluff Lake was decommissioned between 2002 and 2013 and is now in a period of long-term monitoring. Decommissioning plans at each facility are compared to decommissioning activities at Cluff Lake to help validate the estimates.

3.2.7 Data Presentation

In order to protect the confidentiality of data supplied by the operators, facilities will henceforth be referred to anonymously as Mine-Mill A, B, and C.

3.3 Process Chain Analysis

This study employs process chain analysis methodology. Process chain analysis, discussed in detail in Section 2.2.6.1, examines all of the material and energy flows into and out of the product system to assess all life cycle emissions, which include:

- Direct Emissions
    - Process emissions from within the organizational boundary, including:
        - Combustion of fuel in company owned vehicles and equipment
        - Emissions from industrial processes
- Energy Indirect Emissions
    - From generation of imported electricity, heat, or steam consumed by the organization
- Other Indirect GHG Emissions
    - Commuting and business travel by employees
    - Outsourcing (e.g. transportation of products, materials, people, or waste by another organization)
    - Emissions from production of purchased raw or primary materials

- Other

Activities leading to emissions are called **activity factors** whereas the emissions resulting from one unit of activity are called **emission factors**. The overall emissions are calculated by multiplying the activity factor and the emission factor:

$$\boldsymbol{Emissions = Activity\ Factor \times Emission\ Factor} \qquad \textbf{(3.1)}$$

In many cases, activity factors (e.g., diesel consumption, electricity consumption, etc.) are tracked by each facility. Environment Canada (2013) and the IPCC (2014) publish GHG emission factors for direct emissions, but information on emissions from upstream extraction, processing, and transport activities is generally not accessible. To include these emissions, the upstream processes and emission factors are modeled using data from ecoinvent Centre (2013) (subscription required) and/or other peer-reviewed sources.

The data from ecoinvent Centre (2013) are global or regional averages and, as such, have lower accuracy than activities that are modeled directly. Where upstream processes make up more than 2% of total emissions, they are assessed for accuracy and local relevance by comparison with other peer-reviewed data sources.

3.3.1 Emission Factors

Emission factors are available from a number of sources depending on the activity type. For most direct emissions, emission factors are obtained via literature review. For most upstream activities, emission factors are drawn from the ecoinvent v3.0 database.

For major processes, only CO_2, CH_4, and N_2O emissions are considered. The ecoinvent v3.0 database includes a number of minor GHGs in addition to those three.

Tables 3-3 to 3-6 list the unit processes and materials considered in this study along with their emission factors. The stated emission factors are mean values. In the software model, each factor is assigned a confidence interval, usually as a lognormal probability distribution. For activities that have a lot of natural variation or where the data quality is low, the confidence intervals are quite wide. When processes are well understood and consistent, the confidence intervals are narrow.

A number of assumptions have been made to estimate emission factors in the tables. Examples include using the life cycle emission factor of an industrial boiler to estimate the emission factor for a diesel generator. This type of estimate is used only with processes and materials that make a small overall contribution to the calculated results. The encoded uncertainties are adjusted to reflect his lack of precision according to the methodology described in Section 3.5.

This approach to estimating emission factors should not introduce large uncertainties into the model. In most cases, these estimates are extrapolations made on a mass

basis. This means that the resulting estimate will include the correct mass of material and should contain relatively similar proportions of materials (e.g. both pieces are primarily steel with some copper, aluminum, rubber, etc.). The amount of energy used in fabrication and transportation are similarly scaled by mass.

Table 3-3. Major Emission Factors Used in This Study – Energy, Explosives, Processes

Process	Unit	Emission Factors (kg CO_2e/Unit)			Data Source	Notes
		Direct	Indirect	Life Cycle		
Energy						
Propane Consumption	L	1.54	0.42	1.95	(Environment Canada, 2013) - (Direct); (ecoinvent Centre, 2013) - (Indirect)	upstream emissions are primarily from processing and transport
Diesel Consumption	L	2.77	0.55	3.32	(Environment Canada, 2013) - (Direct); (ecoinvent Centre, 2013) - (Indirect)	upstream emissions are primarily from processing
Gasoline Consumption	L	2.3	0.63	2.93	(Environment Canada, 2013) - (Direct); (ecoinvent Centre, 2013) - (Indirect)	upstream emissions are primarily from processing
Electricity Consumption	kWh			0.766		average grid mix 2004-2013 (SaskPower, 2013)
Coal, Gas, Imports, and Other	kWh			0.983	(Saskpower, 2014) - Combustion (OpenEI, 2012) – Other Emissions	77.7% of SaskPower net electricity generation 2004-2013 (SaskPower, 2013); 0.940/0.983 kg CO_2e from combustion
Hydro	kWh			0.0072	(OpenEI, 2012)	19.4% of SaskPower net electricity generation 2004-2013 (SaskPower, 2013)
Wind	kWh			0.0109	(OpenEI, 2012)	2.9% of SaskPower net electricity generation 2004-2013 (SaskPower, 2013)
Explosives					stoichiometry - direct; (ecoinvent Centre, 2013) - indirect	
AN/FO	kg	0.18	8.69	8.87		94.2%wt ammonium nitrate and 5.8%wt fuel oil
AN/FO plus Inert	kg	0.16	6.49	6.65		60%wt ammonium nitrate, 35%wt carbonic acid, and 5%wt fuel oil
Emulsion-Type	kg	0.19	8.04	8.23		79%wt ammonium nitrate, 15%wt sodium nitrate, and 6%wt fuel oil
Process Emissions						
CaCO3 Decomposition	kg $CaCO_3$	0.44		0.44		based on stoichiometry of complete reaction
Na2CO3 Decomposition	kg Na_2CO_3	0.42		0.42		based on stoichiometry of complete reaction
Waste Disposal						
Domestic Solid Waste	kg	0.51		0.51	(Emcon Associates, 1980)	methane emissions only; based on waste composition
Contam. Solid Waste	kg	0.0022-0.018			(Emcon Associates, 1980)	methane emissions only; based on waste composition
Liquid Organic Waste	kg	0-30.4			(Buswell and Mueller, 1952; Riser-Roberts, 1998)	varies based on composition, degree of degradation, degradation processes
Domestic Wastewater	m^3	0.49		0.49	(ecoinvent Centre, 2013)	
Land Use Change	m^2•year	0.233			(Gower et al., 2001)	reduction of existing ecosystem services (carbon sequestration) due to conversion of boreal forest to industrial site; assumes land at disturbed site operates as neither carbon source or sink

Table 3-4. Major Emission Factors Used in This Study – Infrastructure and Stationary Equipment

Process	Unit	Life Cycle Emission Factor (kg CO_2e/Unit)	Data Source	Notes
Infrastructure				
Building, Steel Hall	m^3	47.1	(ecoinvent Centre, 2013)	based on 50mx30mx7m building of steel construction; direct emissions from diesel used during construction; indirect emission primarily from steel, aluminum, brick production
Reinforcing Steel	kg	2.60	(ecoinvent Centre, 2013)	
Aluminum	kg	14.7	(ecoinvent Centre, 2013)	
Brick	kg	0.33	(ecoinvent Centre, 2013)	
Concrete			Cement- (Marceau et al., 2006) Rest-(ecoinvent Centre, 2013)	based on concrete recipes from McArthur River - (Cameco Corporation, 2014d) cement production ~98% of total emissions
Construction	m^3	394		
Raise filling	m^3	360		weighted average for several concrete types used in raise filling
Shotcrete	m^3	396		
Grout	m^3	711		
Mud Slab	m^3	201		
Building Equipment				
Steel, low-alloyed	kg	2.40	(ecoinvent Centre, 2013)	
Chromium steel	kg	1.36	(ecoinvent Centre, 2013)	
Cast Iron	kg	2.21	(ecoinvent Centre, 2013)	
Rubber	kg	3.13	(ecoinvent Centre, 2013)	
Polyethylene Pipe	m	9.99	(ecoinvent Centre, 2013)	1m length of 200mm diameter polyethylene pipe containing 3.15 kg polyethylene, scaled up or down based on relative cross sectional area of pipe material
Industrial Furnace, 1 MW	pc	14,000	(ecoinvent Centre, 2013)	1 MW industrial furnace, 4766 kg; scaled up or down based on relative mass
Boiler, 0.5 MW	pc	16,600	(ecoinvent Centre, 2013)	based on 0.5 MW boiler, 2678 kg, which, in turn is an interpolation of a 0.1 MW boiler and 1 MW industrial furnace (589 kg and 4766 kg respectively); scaled up or down based on relative mass
Diesel Generator, 1.6 MW	pc	60,100	(ecoinvent Centre, 2013)	materials and production emissions based on 1MW Industrial furnace, scaled up by relative mass (20400 kg/4766 kg)
Fan/Blower (720 m^3/hr)	kg	1120	(ecoinvent Centre, 2013)	Based on 182kg blower/heat exchanger, scaled by mass

Table 3-5. Major Emission Factors Used in This Study – Transport and Mobile Equipment

Process	Unit	Life Cycle Emission Factor (kg CO_2e/Unit)	Data Source	Notes
Transport				
Flights, excluding fuel	flt hr	5.16	(ecoinvent Centre, 2013)	Emissions from airport infra: 2.87 kg CO_2e, aircraft production: 2.29 kg CO2e
Jet Fuel	L	3.07	(EPA, 2014; ecoinvent Centre, 2013)	Upstream emissions primarily from processing
Freight, Reagents, Fuel				
5-6 axle van/flatdeck	km	1.60	(Environment Canada, 2013; ecoinvent Centre, 2013)	Emissions from: diesel burning: 1.31 kg CO_2e; rest is vehicle production, maint, and road infra; based on 0.47L/km fuel economy (Dalshaug, 2014)
8 axle flatdeck	km	2.20	(Environment Canada, 2013; ecoinvent Centre, 2013)	Emissions from: diesel burning: 1.46 kg CO_2e; rest is vehicle production, maint, and road infra; based on 0.523L/km fuel economy (Dalshaug, 2014)
9 axle/special configs	km	2.33	(Environment Canada, 2013; ecoinvent Centre, 2013)	Emissions from: diesel burning: 1.87 kg CO_2e; rest is vehicle production, maint, and road infra; based on 0.672L/km fuel economy (Dalshaug, 2014)
Unknown type	T-km	0.0638	(ecoinvent Centre, 2013)	Emissions from diesel burning: 0.0422 kg CO_2e; rest is vehicle production, maint, and road infra; based on 8-axle flatdeck with nominal 34.5T cargo
Vehicles				
Light vehicle	pc	16,100	(ecoinvent Centre, 2013)	production and maintenance; based on 1524 kg light duty vehicle; emission factor scaled up or down based on relative vehicle weight
Light mobile construction	pc	34,300	(ecoinvent Centre, 2013)	production and maintenance; based on 3000 kg tractor; emission factor scaled up or down based on relative vehicle weight
Heavy mobile construction	pc	83,800	(ecoinvent Centre, 2013)	production and maintenance; based on 15,372 kg truck; emission factor scaled up or down based on relative vehicle weight

Table 3-6. Major Emission Factors Used in This Study – Reagents

Reagents	Unit	Life Cycle Emission Factor (kg CO_2e /Unit)	Data Source	Notes
Steel grinding balls	kg	2.05	(ecoinvent Centre, 2013)	
Coagulants, flocculants, anti-scalants	kg	2.14	(ecoinvent Centre, 2013)	Modeled as 'organic chemicals'
Sulfur	kg	0.02	(ecoinvent Centre, 2013)	
Lime	kg	0.10	(ecoinvent Centre, 2013)	
Kerosene	kg	0.59	(ecoinvent Centre, 2013)	
Ammonia	kg	2.07	(ecoinvent Centre, 2013)	
Barium Chloride	kg	2.30	(ecoinvent Centre, 2013)	Modeled as 'inorganic chemicals'
Hydrogen Peroxide	kg	1.30	(ecoinvent Centre, 2013)	
Citric Acid	kg	28.3	(ecoinvent Centre, 2013)	
Potassium Permanganate	kg	1.61	(ecoinvent Centre, 2013)	
Hydrochloric Acid (w/o water)	kg	1.72	(ecoinvent Centre, 2013)	
Isobutanol	kg	3.02	(ecoinvent Centre, 2013)	
Versene (EDTA)	kg	4.31	(ecoinvent Centre, 2013)	
Sodium Hydroxide (w/o water)	kg	1.38	(ecoinvent Centre, 2013)	
Tertiary Amine	kg	3.15	(ecoinvent Centre, 2013)	Modeled as 'triethyl amine'
Nitrogen, Liquid	kg	0.60	(ecoinvent Centre, 2013)	
Sodium Bicarbonate	kg	0.95	(ecoinvent Centre, 2013)	
Sulfuric Acid	kg	0.12	(ecoinvent Centre, 2013)	
Oxygen, Liquid	kg	0.62	(ecoinvent Centre, 2013)	
Quicklime	kg	1.09	(ecoinvent Centre, 2013)	
Iron Sulfate	kg	0.25	(ecoinvent Centre, 2013)	
Magnetite	kg	1.09	(ecoinvent Centre, 2013)	
Kerosene	kg	0.59	(ecoinvent Centre, 2013)	
Sodium Chlorate	kg	4.39	(ecoinvent Centre, 2013)	

3.4 System Modeling

Data are inventoried and analyzed first using spreadsheets. Data for production years are evaluated by year whereas construction, decommissioning, equipment, and infrastructure are considered as blocks of activity.

Emission estimates are valid for the production years where data is available, typically from 2006-2013 for Cameco facilities and 1995-2010 for AREVA.

All of the facilities considered have experienced major production changes throughout their lives. For example, some mills have been in operation longer than the mines from which they currently source their ore. Earlier mining-milling activities involved different mining-milling methods, ore grades, and environmental and safety requirements. No data is currently available to perform a detailed PCA for these early U_3O_8 production periods.

Activities occurring during production years are normalized to U_3O_8 production over that time period. Block activities such as construction and decommissioning are normalized to each facility's lifetime U_3O_8 production.

Future production is, of course, uncertain as it relies on the accuracy of reserve estimates, ore grades, future uranium prices, and other socio-economic factors. It is assessed as follows:

- Base Case
    - Development proceeds as per *Life of Mine Plan* (i.e., 100% of proven and probable reserves are utilized)
- Worst Case
    - 70% of proven and probable reserves are utilized
- Best Case
    - 100% of proven and probable reserves are utilized
    - 50% utilization of measured, indicated, and inferred resources
    - 25% increase in both reserves and resources due to future exploration

Data for reserves and resources is taken from facility technical and environmental reports (Cameco Corporation, 2012a; Cameco Corporation, 2012b; Cameco Corporation, 2013d).

3.4.1 SimaPro

Once the activity inventory is complete, a digital model of each facility is created in SimaPro. SimaPro is a world-leading, ISO 14040-compatible LCA tool which can be used to model products, processes, and facilities in order to understand their impacts.

The software makes it easy to build relationships between processes and to identify key emissions contributors or 'hot-spots'.

In the absence of primary data, SimaPro has an extensive database of secondary data to draw on: the aforementioned ecoinvent database, referenced as ecoinvent Centre (2013). In this research, the ecoinvent database is used to model upstream processes and to estimate emissions in the absence of primary data.

Finally, the software allows uncertainty to be encoded along with each parameter. Using Monte Carlo analysis, the program assesses the overall uncertainty in the result. The ecoinvent parameters are pre-programmed with probability distributions that take into account the regional and temporal variability in the processes they represent as well as quality of the source data.

The details of how the ecoinvent database has been constructed, how uncertainty is treated, and the data validation process are summarized in Weidema et al. (2013).

3.5 Uncertainty

Myriad factors introduce uncertainty to the calculated emission intensity for each facility including data gaps and uncertainty in emission factors.

Where gaps on activity data within the specified operational period exist, the uncertainty in the calculated activity factor is based on the number of years without data and the year-to-year variation of the known data using a lognormal probability distribution. The lognormal distribution was chosen because it provided a good representation of the variation in the available data. Uncertainty in operational activity data is only assessed from 2006 to 2013 for McArthur River, Key Lake, and Rabbit Lake. It is assessed from 1995 to 2010 for McClean Lake.

The 95% confidence interval used to fill data gaps for any given year is calculated as follows:

$$95\%CI = e^{\mu \pm 2\sigma} \tag{3.4}$$

where:

μ = mean of log-transformed data
σ = standard deviation of log-transformed data

Uncertainty is propagated to the calculated total for each activity factor using the root sum of squares method:

$$2\sigma_T = \frac{\sqrt{n(2\sigma)^2}}{n_T} \tag{3.5}$$

where:

$2\sigma_T$ = total uncertainty in summed log-transformed data
n = number of years without data within specified operational period

n_T= total number of years within specified operational period

And the overall confidence interval of the activity factor is:

$$95\%CI = e^{\mu \pm 2\sigma_T} \quad (3.6)$$

The uncertainty factor is encoded in SimaPro as the square of the geometric standard deviation (σ_g^2):

$$\sigma_g^2 = e^{2\sigma_T} \quad (3.7)$$

This method considers variation year-by-year and ignores changes in operation, production volumes, and ore grade. Inclusion of these other parameters would likely reduce the uncertainty in the result, but the calculation is not straightforward and, as will be shown in Section 4.4, is not required to produce a reasonably precise result.

When the above method cannot be applied (e.g., emission factors, infrastructure and equipment), uncertainty is assessed based on the methodology used in the SimaPro software and ecoinvent database, described in Weidema et al. (2013). The methodology is also endorsed in the Greenhouse Gas Protocol, which itself provides reporting standards, sector guidance, and calculation tools for quantifying and reporting GHG emissions for companies and organizations around the world (WRI/WBCSD, 2014b).

The probability distribution of each parameter is defined by assessing and combining six categories of uncertainty:

1. Basic Uncertainty
2. Reliability
3. Completeness
4. Temporal Correlation
5. Geographical Correlation
6. Further Technological Correlation

The first, *Basic Uncertainty*, is applied in absence of sampled data. It is modeled as a lognormal probably distribution with the square of the geometric standard deviation ranging from 1 to 3 depending on the inherent uncertainty in the type of data as per Table 3-7.

Table 3-7. Basic Uncertainty Factors from Weidema et al. (2013)

	Basic Uncertainty Factor $U_1 = \sigma_{g1}^2$
Demand of:	
Energy	1.05
Materials	1.05
Transport services	2.00
Infrastructure	3.00
Pollutants emitted to air:	
CO_2	1.05
CH_4	1.50
N_2O	1.50

The other five sources of uncertainty are applied using the indicator score definitions shown in Figure 3-2 and their associated uncertainty factors, listed in Table 3-8. Figure 3-2 only shows definitions for the extreme values of each indicator score. Full definitions are available in Weidema et al. (2013).

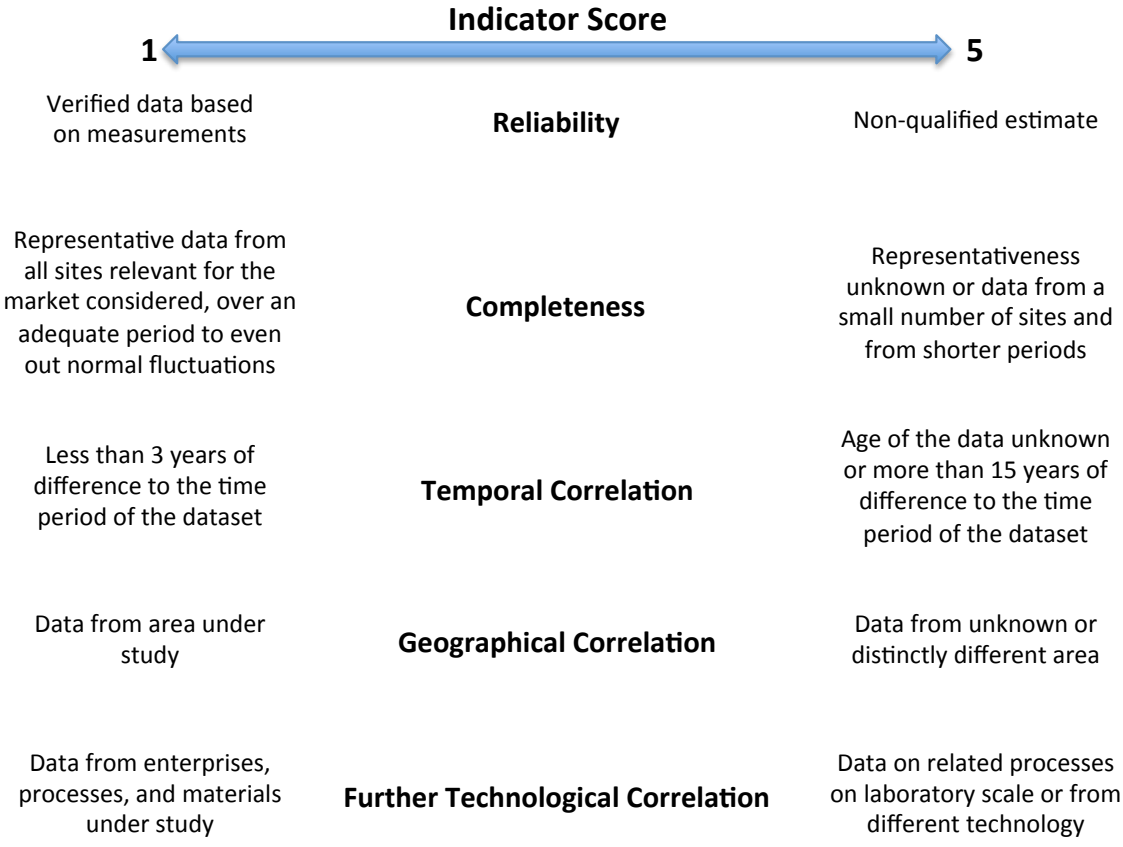

Figure 3-2 Indicator Score Range and Definitions Used in Assessing Uncertainty Factors (Weidema et al., 2013)

Table 3-8. Elemental Factors Used in Assessing Total Uncertainty Factor. Reported as square of geometric standard deviation (Weidema et al., 2013)

Indicator Score	1	2	3	4	5
Reliability	1.00	1.05	1.10	1.20	1.50
Completeness	1.00	1.02	1.05	1.10	1.20
Temporal Correlation	1.00	1.03	1.10	1.20	1.50
Geographical Correlation	1.00	1.01	1.02	1.05	1.20
Technological Correlation	1.00	1.05	1.20	1.50	2.00

Total uncertainty is calculated using the following formula:

$$\sigma_g^2 = \exp\sqrt{[\ln(U_1)]^2 + [\ln(U_2)]^2 + [\ln(U_3)]^2 + [\ln(U_4)]^2 + [\ln(U_5)]^2 + [\ln(U_6)]^2} \quad (3.8)$$

where:

U_1= basic uncertainty factor
U_2= uncertainty factor of reliability
U_3= uncertainty factor of completeness
U_4= uncertainty factor of temporal correlation
U_5= uncertainty factor of geographical correlation
U_6= uncertainty factor of further technological correlation; and
U_n = the square of the element's geometric standard deviation.

The uncertainties in each parameter are propagated to the total result using the Monte Carlo method. In this method, each parameter is varied randomly within its probability distribution. The total result is calculated repeatedly based on the assigned values of its constituent parameters. With repeated calculation, a probability distribution for the total emerges. In this study, each total was calculated based on 5000 runs and a 95% confidence interval is reported.

Due to the qualitative aspect of this methodology, a confidence interval calculated in this way is not a precise value. Rather, it provides an indication of the overall uncertainty in the results and the relative uncertainty in the processes that make up the system model.

Section 4.4 includes an analysis of the system model's uncertainty including a sensitivity analysis based on the most influential activities.

3.6 Unit Conversion

As discussed in Section 2.5.1.4, the emission estimates from the mining-milling phase of nuclear power range from 0.05 to 24.73 kg CO_2e/kWh with a median value of 1.5 kg CO_2e/kWh based on 13 data points. As mentioned, not all studies disaggregate results by life cycle phase which is why there are not more data points.

The units (kg CO_2e/kWh) for emission intensity (I_{kWh}) used in these studies are not directly comparable to the results of the current study, which are presented in kg CO_2e/kg U_3O_8 ($I_{U_3O_8}$). The final value will be different depending on the type of reactor used and its operating parameters, namely burn-up (B) and thermal efficiency (η_{th}). The amount of natural uranium required to produce a unit of enriched uranium (η_{enr}) is also important.

A unit conversion can be performed as follows:

$$I_{kWh} = \frac{I_{U_3O_8}}{B \times \eta_{th} \times \eta_{enr}} \quad (3.9)$$

Using an example in Fthenakis and Kim (2007), a light water reactor (LWR) may burn-up at B = 42 MW_{th}d/kg U, operate at η_{th} = 0.3, and require the enrichment of 7 kg U to generate a 1 kg enriched U fuel with 3.8% U-235 content (η_{enr}=0.1429). Using these parameters yields the following conversion factor:

$$\frac{I_{kWh}}{I_{U_3O_8}} = 2.7 \times 10^{-5} \ \frac{kg\ U_3O_8}{kWh}$$

Heavy water reactors (HWR) burn-up at approximately B = 8 MW_{th}d/kg U, have a similar thermal efficiency, and do not require uranium enrichment (η_{enr}=1) (Andseta et al., 1998; Wilk, 2013). This yields an emission intensity factor of $2.0 \times 10^{-5} \ \frac{kg\ U_3O_8}{kWh}$.

Note that these calculations required the conversion of mass U to U_3O_8 and MWd to kWh. When converting between units in Section 4, this is the formulation and these are the factors that will be used. The factors and assumptions used in the above calculation are not always included in the studies reviewed.

4. Results and Discussion

4.1 Emissions Intensities

Figure 4-1 shows the life cycle analysis greenhouse gas (GHG) emission intensity results plotted against ore grade for each facility including the 95% confidence intervals.

As predicted, life cycle GHG emissions intensities decrease as ore grades increase. The empirical relationship between ore grade and GHG emission intensity proposed in Mudd (2014) is shown and is consistent with the mine-mill results from this study. Note that the Mudd (2014) curve is generated, in part, with data from the facilities included in this study (see Section 2.6 for more details).

Figure 4-1 also shows the production-weighted average emissions intensity for U_3O_8 produced in SK. Weights are calculated based on relative U_3O_8 production volumes at Mill-A, B, and C from 2006 to 2010. All facilities were operating at normal capacity during this time period.

When comparing life cycle GHG emissions from electricity generation, all of the estimates included in the literature review (Section 2.5) report emission intensity in terms of g CO_2e/kWh. Figure 4-2 shows the GHG emission intensity of the uranium mining-milling phase in these units using a conversion factor of $2.7\times10^{-5}\ \frac{kg\ U_3O_8}{kWh}$ as developed for light water reactors in Section 3.6. The production-weighted average result from this study is plotted against the range of values from the eight studies presented in Section 2.5 that reported emissions in the mining-milling phase for uranium from outside of SK.

The LWR conversion factor was chosen for the figure because that is the reactor technology most commonly reported in the literature reviewed. If the same calculation were made using the conversion factor of $2.0\times10^{-5}\ \frac{kg\ U_3O_8}{kWh}$ as developed for heavy water reactors, the resultant emission intensity would be 26% lower.

As shown in Figure 4-2, the results for the mining-milling of uranium in SK are low compared to estimates for uranium from outside of SK.

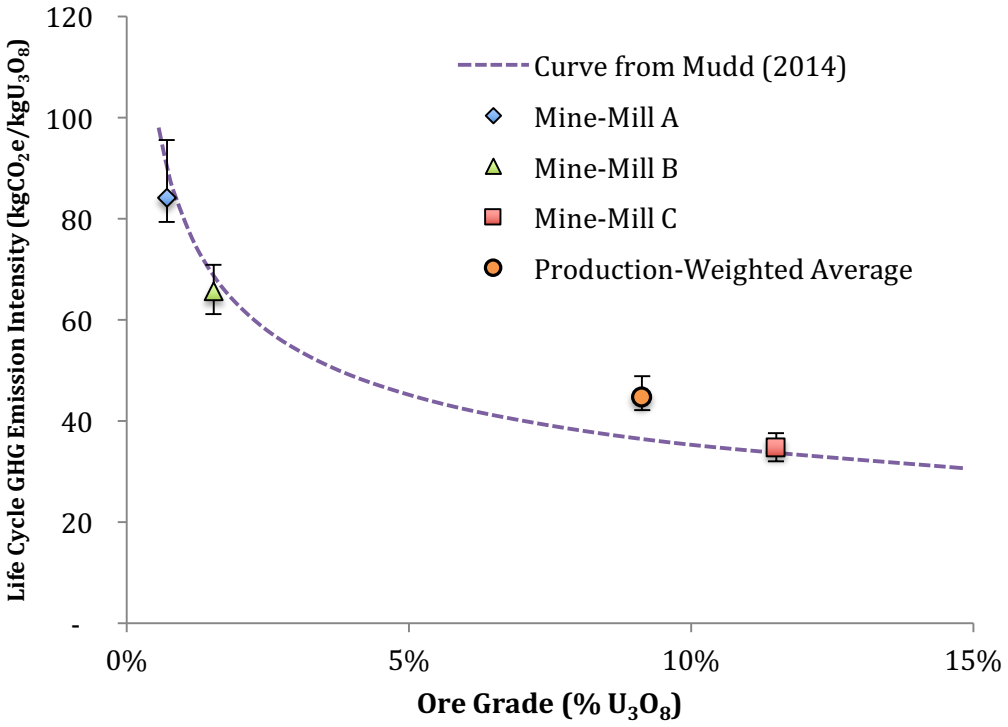

Figure 4-1. Detail for Life Cycle GHG Emission Intensity vs. Ore Grade. Bars indicate 95% confidence interval.

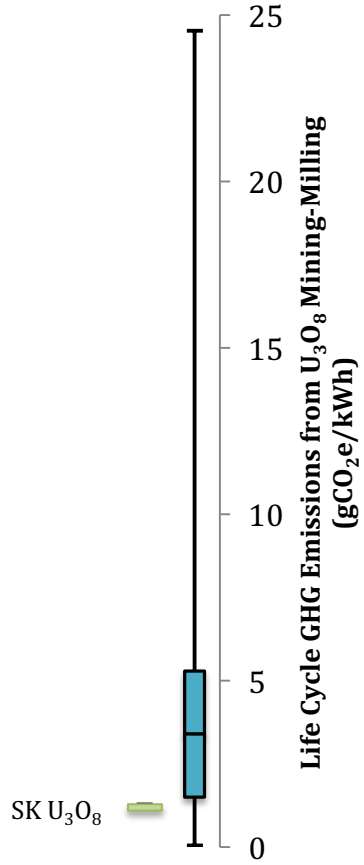

Figure 4-2. – Life Cycle GHG Emissions from Uranium Mining-Milling in SK vs. Other Global Estimates in Meta-Analyses by Beerten et al. (2009), Fthenakis and Kim (2007), and Sovacool (2008). Boxes represent 2nd and 3rd quartiles. Whiskers represent total range

4.2 Processes

Table 4-1 shows the GHG emission intensity for each facility by process. Figure 4-3 shows the relative contributions of each process for each mine-mill and the production-weighted average result. The results are generally similar for Mine-Mill A, B, and C.

Table 4-1. GHG Emissions Intensity (kg CO_2e/kg U_3O_8) by Facility and Process

	Mine-Mill A	Mine-Mill B	Mine-Mill C	Production-Weighted Average
Average Ore Grade (% U_3O_8)	0.71	1.54	11.5	9.12
Process				
Electricity	36.4	18.2	13.6	17.6
Propane	19.6	15.6	7.1	9.6
Diesel	6.4	7.3	2.4	3.3
Reagent Consumption	3.1	9.0	1.8	2.5
Construction, Infrastructure and Equipment	3.7	0.5**	1.9	2.1
Land Use Change	2.8	2.2	1.4	1.7
Truck Transport	3.1	2.6	1.3	1.6
Passenger Commute	3.0	2.3	1.1	1.5
Concrete Production*	0.4	0.0	1.7	1.4
Use of Explosives	1.0	6.5	0.1	0.7
Process Emissions	0.7	0.4	0.7	0.7
Decommissioning	1.8	0.1	0.4	0.6
Gasoline	0.7	0.6	0.3	0.4
Other	1.7	0.4	0.9	1.0
Total (kg CO_2e/kg U_3O_8)	**84**	**66**	**35**	**45**
Total (g CO_2e/kWh - LWR)	**2.3**	**1.8**	**0.94**	**1.2**

*concrete used in mining, not in infrastructure

** some emissions from construction for Mine-Mill B captured in other processes

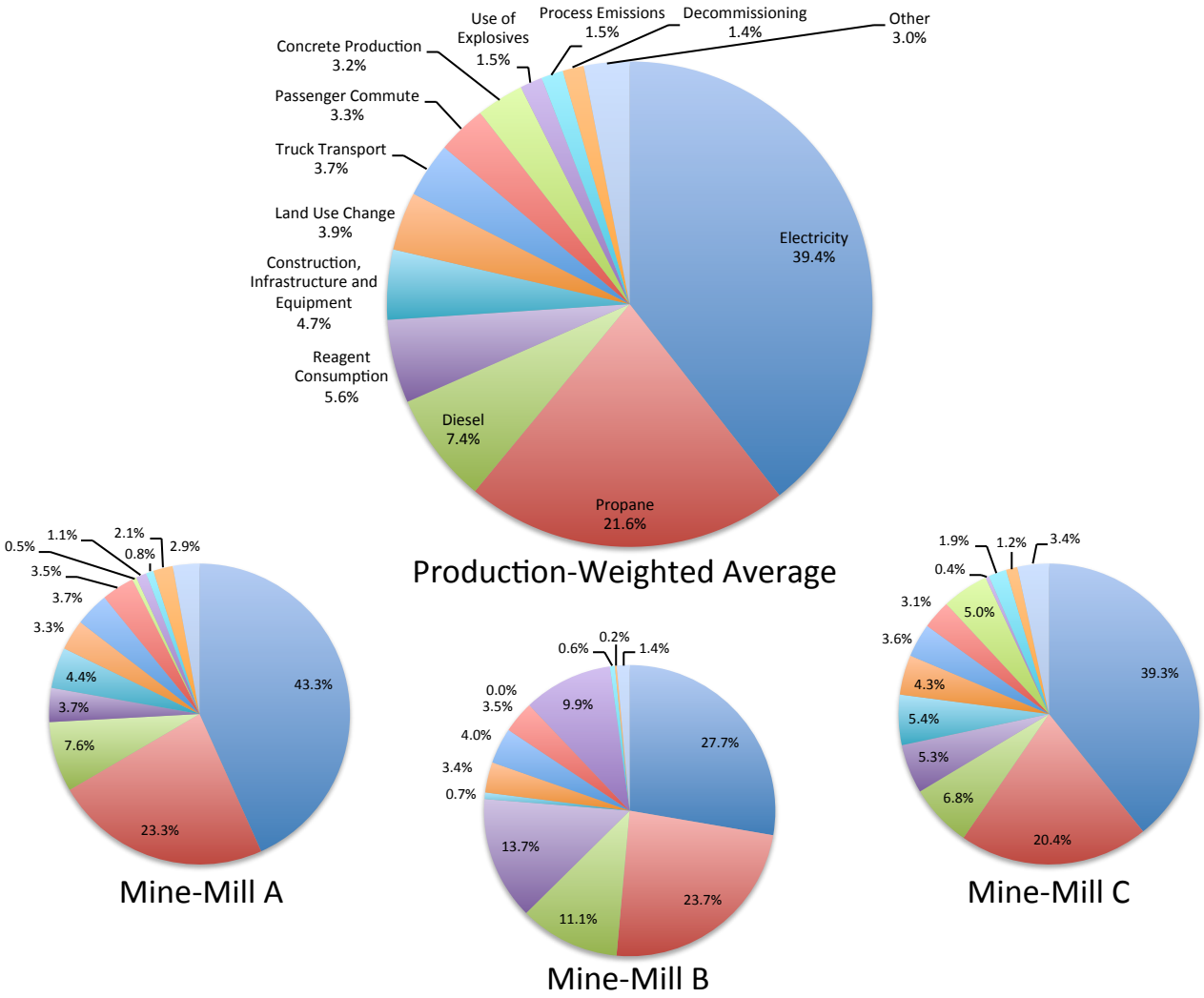

Figure 4-3. Relative Contribution by Process for Each Facility and Production-Weighted Average. Processes are ordered and colored in the same way for each pie. Energy consumption (electricity, propane, and diesel) as shown includes only direct consumption during site operation. Energy consumption during construction, decommissioning, or in upstream processes are included as part of their respective categories.

GHG emissions arise primarily from energy consumption at site during operation (i.e., electricity, propane, diesel, and gasoline [shown in table, listed as 'Other' in Figure 4-3]) which, combined, account for 69% of the total on average. This includes direct emissions from fuel consumption, direct emissions at the power plants that provide electricity, and upstream and embodied emissions for those activities.

Due to the remote nature of the sites, all personnel are flown to and from each facility, and most freight travels in excess of 700 km by road each way. Transportation of materials and personnel accounts for 7.0% of the emissions total. Note that upstream transportation for fuels, reagents, equipment, etc. are captured in their respective categories and not included in the *Truck Transport* category.

The consumption of reagents during mining-milling contributes 5.6% to the emissions total, mostly embodied in ammonia, lime, and hydrogen peroxide which account for 3.6%.

The *Construction, Infrastructure, and Equipment* category includes emissions embodied in materials as well as the energy required for construction and the transportation of materials and personnel to support those activities. There are non-trivial emissions from these activities (4.7%), but, in the context of the full life cycle, they are a small component. The same is true for decommissioning activities.

Emissions from *Land Use Change* (3.9%) arise from the reduction in existing ecosystem services. The boreal forest, where these facilities are located, operates as a carbon sink under normal conditions (Gower et al., 2001). It is assumed that, during the disturbance period, the land functions as neither a carbon sink nor a source.

Concrete consumption also makes a large contribution to the emission total (3.2%) due in part to the energy-intensive pyroprocess used upstream in the cement-production process (Marceau et al., 2006).

Non-combustion process emissions (1.5%) arise from the liberation of CO_2 from calcium carbonate in ore and sodium bicarbonate added during the milling process. It is assumed that these carbonates react completely upon exposure to the high-strength acid used during milling.

Other includes gasoline consumption, methane emissions from the landfilled solid waste, methane and carbon dioxide emissions from landfilled organic liquid wastes, methane emissions from domestic wastewater, direct and indirect emissions from natural gas and electricity consumption at corporate headquarters, and direct emissions from exploration activities.

4.3 Comparison to Literature Review

It was hypothesized that the life cycle GHG emissions intensity for uranium from SK would be lower than values reported for other facilities around the world due to the very high ore grades (averaging 9.12% U_3O_8 over the study period) processed at SK facilities compared to other facilities worldwide (typically 0.1-0.2% U_3O_8). The production-weighted average result is indeed lower than six of the eight estimates found in meta-analyses by Fthenakis and Kim (2007), Beerten et al. (2009), and Sovacool (2008) for ore from outside of Saskatchewan, but it is higher than the other two. Unfortunately, the papers that detail the analysis for the mine-mills used for these two lower estimates are written in Japanese (Tokimatsu et al., 2006) and Dutch (Torfs et al., 1998) and are not readily available. Without this information, it is not possible to explore the reasons for these differences.

The other three studies included in the literature review that report emisisons from mining-milling are based on U_3O_8 sourced from Canada (where the vast majority of uranium comes from SK). Hondo (2005) reports a very similar estimate to the present study but does not provide an ore grade. Fthenakis and Kim (2007) and Andseta et al. (1998) report lower emission estimates based on similar ore grades.

Figure 4-4 shows the life cycle analysis greenhouse gas (GHG) emission intensity results plotted against ore grade for each facility alongside emission estimates from the literature review, Section 2.5.

In Figure 4-4, the lowest life cycle GHG emission intensity shown is 3.7 kg CO_2e/kg U_3O_8 @ 12.7% U_3O_8. It is taken from Fthenakis and Kim (2007) and is a `best-case` scenario based on the estimated energy consumption required in exploration, mining, and milling of Canadian uranium. The methodology is not clearly stated in the paper, but it appears that only energy consumption is assessed and that the `best-case` scenario is an extrapolation from the `base-case`.

Since Canadian ore is fully represented in the results of this thesis and the results of this thesis are higher (35-84 kg CO_2e/kg U_3O_8), Fthenakis and Kim's results appear to be underestimated. This may be attributed to the use of a smaller study boundary for the mining-milling phase, and the use of extrapolated estimates rather than actual production numbers.

The `base-case` scenario from Fthenakis and Kim (2007) reports a result of 63 kg CO2e/kg U_3O_8 @ 0.2% U_3O_8. As above, an incomplete boundary has likely led to underestimation.

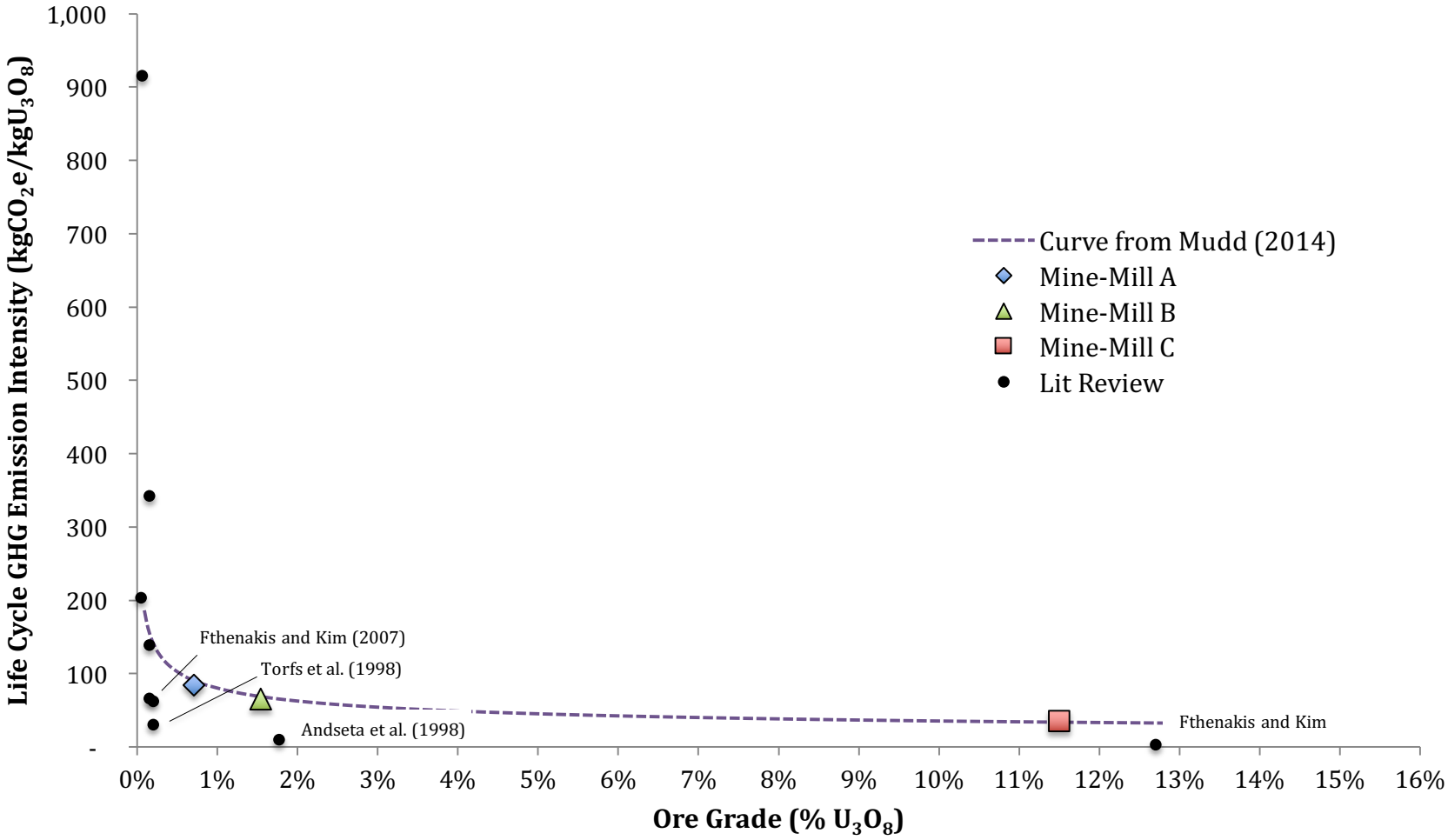

Figure 4-4. Life Cycle GHG Emission Intensity vs. Ore Grade. Black dots are results from studies included in the literature review, Table 2-3, where data for the mining-milling phase and ore grade is available. Conversion to kg CO_2e/kg U_3O_8 from kg CO_2e/kWh is made using conversion factors established in Section 3.6.

Andseta et al. (1998) report a result of 10 kg CO_2e/kg U_3O_8 @ 1.77% U_3O_8. This is a weighted average based on production at Key Lake, Rabbit Lake, and Cluff Lake in 1996 (Andseta et al., 1998). Like Fthenakis and Kim (2007), the evaluation of the mining-milling phase is not a complete life cycle analysis. They considered only energy consumption and reagent oxidation (Andseta et al., 1998). The study assumes that the electricity at two of the three facilities is generated at hydroelectric power stations with an emission factor of zero and that the third facility is powered exclusively by diesel generators. All Scope 3 emissions are omitted.

The production-weighted average GHG emissions intensity presented in this thesis is higher than in Andseta et al. (1998) (45 vs. 10 kg CO_2e/kg U_3O_8), includes two of the same facilities, and has a much higher average ore grade (9.12% vs. 1.77% U_3O_8) Even if emissions from electricity consumption are neglected entirely, the life cycle emissions presented in this thesis are more than double the results in Andseta et al. (1998). Given the widely divergent results and the rigor of the present study, it is reasonable to conclude that Andseta et al. (1998) have underestimated the emissions from the mining-milling of SK uranium.

Underestimations by Fthenakis and Kim (2007), Andseta et al. (1998), and others likely arise from a known shortcoming of the process chain analysis methodology: the systematic underestimation of results due to truncation at the system boundary (Weisser, 2007).

This study addresses the problem of underestimation by including a very large and inclusive system boundary and low cut-off criteria of 0.1%. Further, the use of the SimaPro modeling software and the ecoinvent v3.0 life cycle database allowed the inclusion of many processes failing this cut-off criteria. As a result, the systematic errors due to the use of Process Chain Analysis are minimized.

The magnitude of the discrepancy in the results of this study compared to those found in the literature review evidences the importance of using consistent boundaries when comparing LCA studies. Many studies include only energy-related emissions during operation in their analyses. If the same approach were taken here, the emission estimate would be at least 30% lower.

4.4 Uncertainty Analysis

Uncertainty in the data are assessed as described in Section 3.5. The 95% confidence interval for the production-weighted average result is shown in Figure 4-5. The figure also presents the result's sensitivity to uncertainty in the most influential processes. For each horizontal bar in the figure, values associated with the listed process were varied within their probability distributions while all values associated with other processes were held constant.

Even though the system model included a number of elements based on partial data, assumptions, and estimates, the overall uncertainty in the result is reasonably low, -2.6/+4.1 kg CO_2e/kg U_3O_8 (-5.7%/+9.2%). This is because data for the largest emission-driving activities (e.g., energy consumption, transport) was available and the associated emissions factors were well understood, especially in the case of fuel combustion.

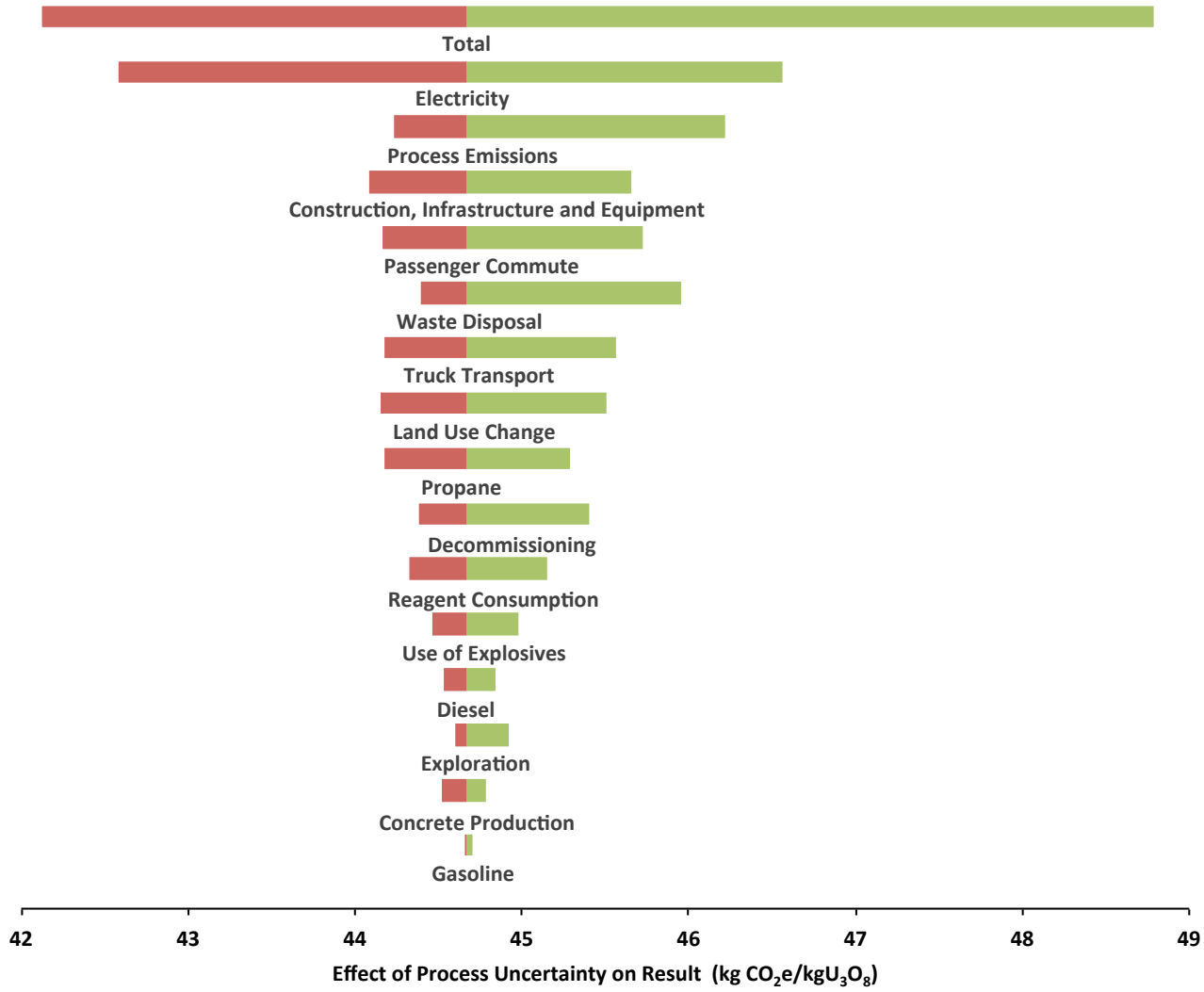

Figure 4-5. Confidence Interval (95%) of Production-Weighted Average Result and Analysis of Model Sensitivity to Most Influential Processes

The largest contributor to overall uncertainty is *Electricity Consumption*, primarily because it is the single largest source of emissions. While there is negligible uncertainty in the amount of electricity consumed, the same is not true for the emission factors used. It is unlikely that this source of uncertainty could be significantly reduced without a great deal of effort including collaboration with the utility as it would require a detailed life cycle analysis of the specific power plants used in SK.

The second-largest element of uncertainty is in *Process Emissions*, specifically those arising from the liberation of CO_2 from carbonates in ore when exposed to acid during the milling process. Emissions from this process are directly related to the carbonate content of the ore for which there is relatively little data.

For transportation, fuel economy data is based on fleet-wide averages which may systematically under- or over-estimate fuel consumption during the trips that trucks and planes make when servicing the mines and mills. Uncertainty in *Truck Transport* also arises from the simplifying assumption that all trips depart from Saskatoon.

For *Construction, Infrastructure and Equipment*, uncertainty arises from data gaps in construction-related energy consumption, the mass of materials used in infrastructure, and the specific emission factor for each material.

Emission intensity from *Construction, Infrastructure and Equipment* is also affected by the estimated future U_3O_8 production at each facility. As the facility operates for longer and processes more U_3O_8, the relative emissions from *Construction, Infrastructure, and Equipment* per kg U_3O_8 decrease. Uncertainty in future production values is assessed as described in Section 3.4 and makes up a portion of the uncertainty in this category.

Waste Disposal includes methane emissions from contaminated and domestic solid waste deposited in site landfills, as well as carbon dioxide and methane emissions released from liquid organic waste. The latter is the largest source of uncertainty. There are a number of potential fates for the chemicals including long-term storage in the soil or degradation. The organics may be degraded aerobically or anaerobically and, depending on the process, different amounts of CO_2 and CH_4 will be released (Riser-Roberts, 1998). From those releases, the CO_2 may be emitted to the atmosphere, or mineralized and stored in the soil. The CH_4 may be released to the atmosphere or further oxidized to CO_2 (Riser-Roberts, 1998). The actual fate will be determined by environmental factors. A detailed model of the environmental fate of the liquid organic waste stream is beyond the scope of this study. The uncertainty profile for this emission source varies from zero emissions (all waste are stored in soil indefinitely) to complete methanogenesis.

Estimated loss of carbon sequestration capacity due to land use change depends on the size of the disturbed area, the time between construction and reclamation, and

the pre-existing net primary carbon sequestration rate (net primary productivity, NPP) in the region. The latter value is the least understood. The nominal value of 233 g/m^2year is based on evergreen conifer stands near the southern edge of the boreal forest at latitudes 53.99 and 55.99 N with mean annual temperatures of -1.1 and -4.6°C respectively and mean annual precipitation of 405 and 536 respectively (Gower et al., 2001).

The facilities included in this study, while located in the boreal forest, are more than 130 km further north (57.25N or higher) with similar annual temperatures and precipitation levels (-2.3°C and 482.5mm respectively) as the locations specified in Gower et al. (2001) (Government of Canada, 2015). Gower et al. (2001) find a strong negative correlation between NPP and latitude. For this reason, the chosen emission factor, NPP = 233 g/m^2year, is likely a conservative estimate.

Uncertainty for most other elements occur far upstream in the system model, due to natural variability in manufacturing processes, transportation distances, etc.

Given the low overall uncertainty in the result and the effort required to achieve better precision, it may be impractical and unnecessary to attempt further uncertainty reduction.

4.5 Scenario Analysis: Electricity from Hydro

Northern SK is powered exclusively by two SaskPower hydroelectric generation facilities and an interconnection with Manitoba Hydro, which generates approximately 96% of its power from hydroelectricity (Manitoba Hydro, 2014; SaskPower, 2013). There is no direct connection intra-provincial grid connection between northern and southern SK (SaskPower, 2013). This is illustrated in Figure 4-6.

Combined, uranium mines in northern SK used more than 60% of all the power produced at SaskPower's northern hydroelectric stations between 2006-2013 (assuming a 40% capacity factor for each power station) (AREVA Resources Canada Inc., 2014a; Cameco Corporation, 2014d; SaskPower, 2013).

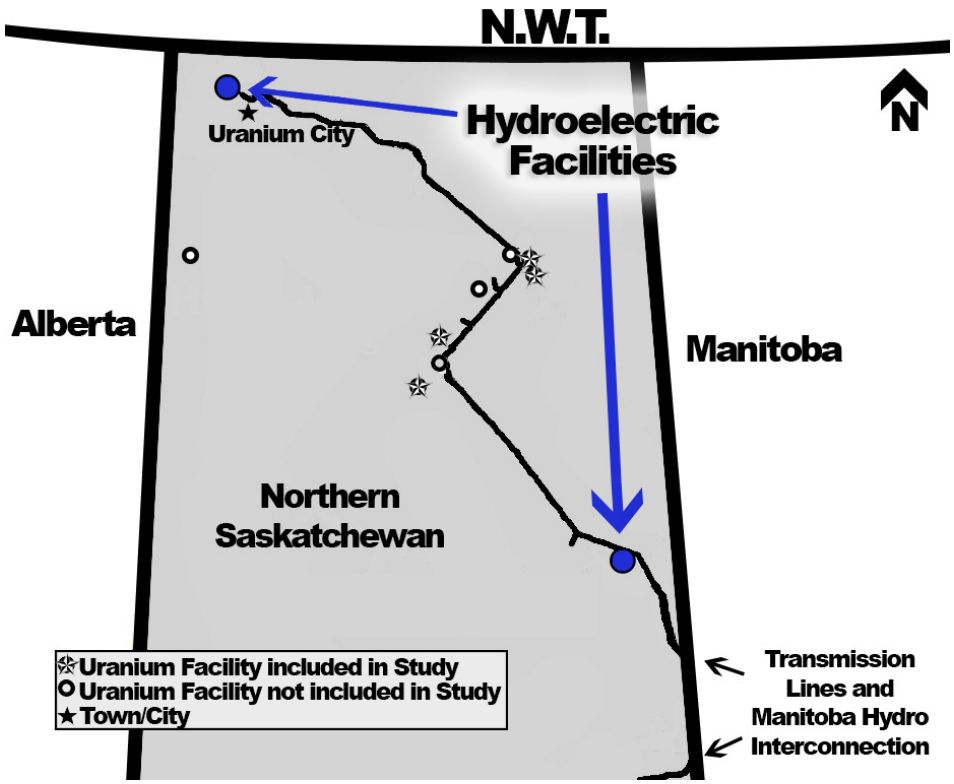

Figure 4-6. Location of Uranium Mines in Northern SK in Relation to Hydroelectric Facilities. Data from SaskPower (2013).

SaskPower provides GHG emission factors for its operations as a whole, which span the province of SK, and does not differentiate their emission factors by region. But, for the reasons stated above, there may be an argument for attributing all electricity generation for facilities included in this study to hydropower.

Given that electricity makes such a large contribution to total energy consumption at the mine-mills, using the hydroelectric emission factor (7.2 g CO_2e/kWh) rather than the provincial average (766 g CO_2e/kWh) will dramatically reduce the emissions estimate for each facility as illustrated in Figure 4-7. Results are summarized in Table 4-2 and the relative contribution of each process is shown in Figure 4-8.

Table 4-2. GHG Emission Intensity based on SaskPower Average Emission Factor vs. Hydroelectric Emission Factor

Facility	GHG Emission Intensity (kg CO_2e/kg U_3O_8) Based on electricity from:	
	SaskPower Average	Hydroelectric Only
Mine-Mill A	84	45
Mine-Mill B	66	48
Mine-Mill C	35	20
Production-Weighted Average	45 (+4/-3)	26 (+1/-3)

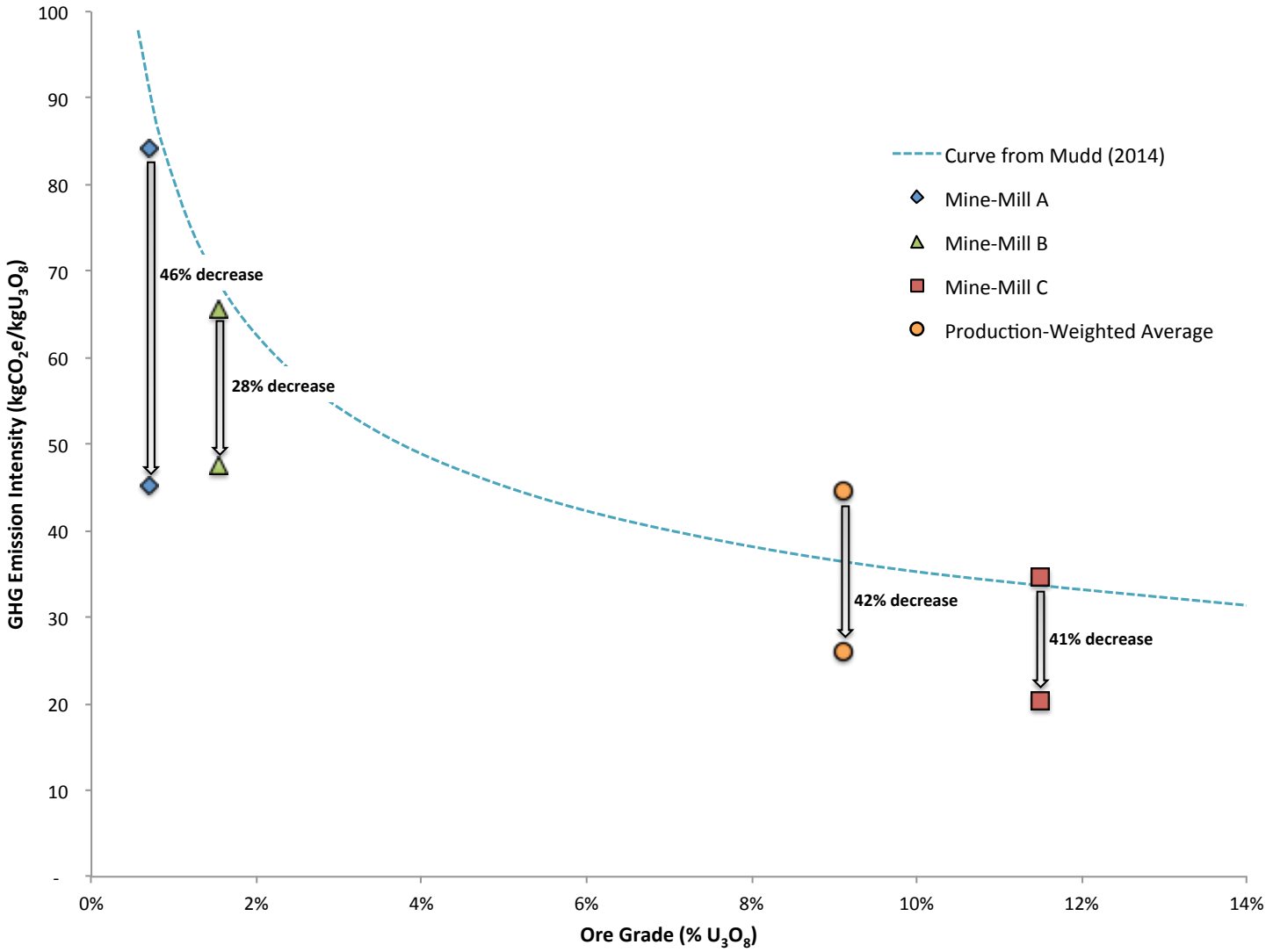

Figure 4-7. Change in GHG Emission Intensity by Using Hydroelectric Emission Factor Rather than Average SaskPower Grid Mix

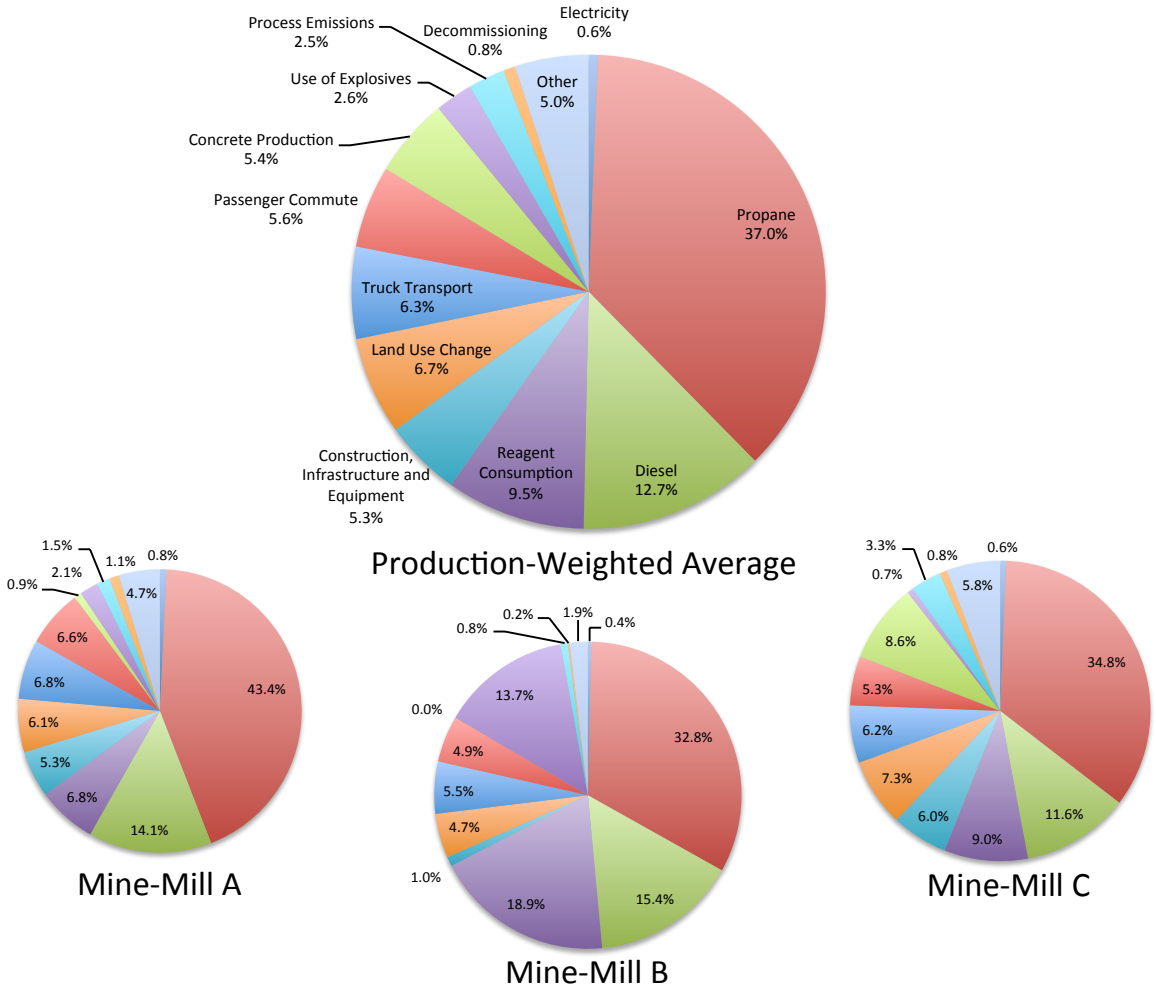

Figure 4-8. Relative Contribution by Process for Production-Weighted Average Using Hydroelectric Emission Factor rather than Average SaskPower Grid Mix. Processes are ordered and colored in the same way for each pie. Energy consumption (electricity, propane, and diesel) as shown includes only direct consumption during site operation. Energy consumption during construction, decommissioning, or in upstream processes are included as part of their respective categories.

In the hydroelectric scenario, electricity goes from being the dominant emission source to one of the smallest. Electricity's contribution to the emissions total shrinks from 40% to less than 1%. As a secondary effect, the low hydroelectric GHG emission factor reduces the emission estimate for construction and decommissioning during the time period that electricity is supplied from the grid for those activities.

Clearly, the choice of GHG emission factor for electricity is of great importance in developing a clear and accurate life cycle emissions estimate for the mining-milling of uranium in SK. This choice, however, is somewhat more complex than presented here. For example, if northern consumers report emissions based on a lower northern regional emission factor, southern consumers should report emissions based on a higher southern regional emission factor in order for GHG emissions throughout the province to be accurately represented. A review of common practices in this regard is outside of the scope of this study, but it is probably safe to assume that southern consumers would prefer to report emissions based upon the lower provincial average emission factor, given the choice.

4.6 GHG Emissions from Mining-Milling in Context of Nuclear Fuel Cycle and Other Technologies

It is important to understand GHG emissions from uranium mining-milling in the context of the full nuclear fuel cycle and compared to other electricity-generating technologies. Figure 4-9 shows emission estimates for the nuclear fuel cycle as presented in the literature review, Section 2.5, and compares those results to a scenario where the SK-produced U_3O_8 fuels the reactors. Selected descriptive statistics are listed in Table 4-3.

As shown, the influence is very small, yielding a 1.5% increase to the median estimate using the production-weighted average results, or a 1.0% decrease under the assumption that all electricity comes from hydroelectric plants.

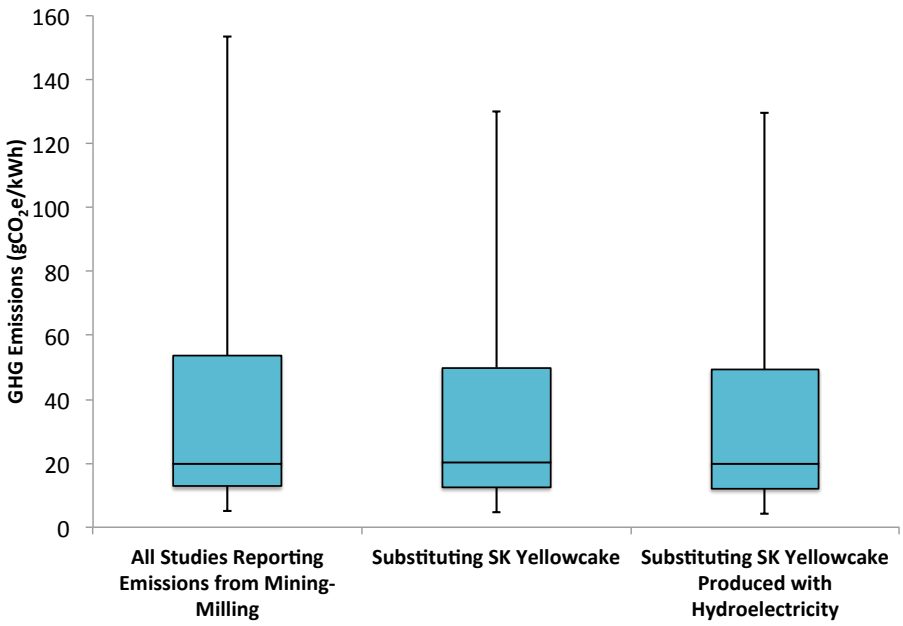

Figure 4-9. GHG Emission Estimates from Full Nuclear Fuel Cycle as Reported in the Literature vs. Results Using Saskatchewan U_3O_8. Data from (Fthenakis and Kim, 2007; Sovacool, 2008; Beerten et al., 2009). Boxes represent 2nd and 3rd quartile. Whiskers represent maximum and minimum values.

Table 4-3. Selected Descriptive Statistics for GHG Emission Estimates from Full Nuclear Fuel Cycle as Reported in the Literature vs. Results Using Saskatchewan U_3O_8

	All Studies Reporting Emissions from Mining-Milling	Substituting Saskatchewan Yellowcake	Substituting Saskatchewan Yellowcake Produced with Hydroelectricity
Median	20.0	20.3	19.8
Mean	40.2	37.4	37.0
IQR	41.0	37.4	37.3
Range (Max – Min)	148.4	125.2	125.2
N	14	14	14

For the facilities included in this thesis, the mining-milling phase make a small but non-negligible contribution to GHG emissions from the full nuclear power fuel cycle: 1.2 g CO_2e/kWh or 10% of the full LWR-life cycle median estimate in Warner and Heath (2012) of 12 g CO_2e/kWh. By comparison, the mining of coal releases 9 g CO_2e/kWh and the production and distribution of natural gas releases 125 g CO_2e/kWh (Spath et al., 1999; Spath and Mann, 2000). This comparison is complicated by the fact that yellowcake requires further processing (e.g., refining, enrichment) to be power plant-ready. These activities are outside of the scope of analysis in this study, but could contribute an additional 0.65-25.8 g CO_2e/kWh (Fthenakis and Kim, 2007; Sovacool, 2008; Beerten et al., 2009).

Note that the GHG emission estimate for the nuclear fuel cycle used above is from Warner and Heath (2012) instead of the data sources for Figure 4-9 and Table 4-3. The former is based on 99 independent estimates of GHG emissions, all of which have been through a harmonization process, compared to the latter which includes 14

unharmonized estimates. The data used in that figure and table were the only sources from which emissions from the mining-milling phase could be extracted.

As a concluding statement in his analysis of global uranium trends, Gavin Mudd states "for mining to begin to materially affect the carbon intensity of nuclear electricity, the carbon intensity of mining would have to increase at least a hundred fold or more (Mudd, 2014)." This statement is true in the sense that emissions from mining-milling would have to increase considerably for emissions from the nuclear fuel cycle to be comparable to those from fossil fuel technologies, but an increase of that magnitude would make nuclear power less emissions-competitive with renewable alternatives.

Fortunately, emissions of this magnitude are not observed. In Mudd (2014), the highest reported emission intensities occur at Langer Heinrich and Rössing uranium mines in Namibia which have comparatively low ore grades, 0.03-0.09%. The highest reported emissions at these sites are approximately 85 and 97 kg CO_2e/kg U_3O_8 respectively (1.8 and 2.1 g CO_2e/kWh) (Mudd, 2014). These reported emissions are calculated on an annual basis and do not appear to include emissions from construction, decommissioning, exploration, or other Scope 3 emissions, the inclusion of which would necessarily increase the estimate. However, the results of the present study suggest that emissions from energy consumption during operation make up the majority of life cycle GHG emissions and these are the ones that are reported in Mudd (2014).

4.7 Application of Results to Other Facilities

The results of this study apply specifically to those facilities included in the analysis. However, where facilities exist under similar conditions as those included, the results may be useful in formulating a first approximation of their life cycle GHG emissions.

This approximation will require a critical review of the processes that contribute to GHG emissions and their associated emissions factors. These can be compared against the process-by-process results shown in Table 4-1. The values given in Table 4-1 can then be modified to represent the new facility.

To correct for ore grade for a given process, interpolate between the given data points for each facility using either a linear interpolation between the two facilities with immediately higher and lower ore grades, or using a power regression based on all three facilities.

Using electricity as an example, Mine-Mill A, B, and C have emissions of 36.4, 18.2, and 13.6 kg CO_2e/kg U_3O_8 at ore grades of 0.71, 1.54, and 11.5% U_3O_8 respectively. A power regression for emissions intensity from electricity based on these results takes the following form:

$$6.44(ore\ grade)^{-0.312}\ R = 0.785$$

where ore grade is expressed as % U_3O_8 and the result is given as kg CO_2e/kg U_3O_8.

If the new facility has an ore grade of 1.2% U_3O_8, the power regression yields an emission intensity from electricity of 25.6 kg CO_2e/kg U_3O_8. Alternatively, using a linear interpolation between the Mine-Mill A and B yields 28.9 kg CO_2e/kg U_3O_8. The decision to use either the linear interpolation or power regression should be determined on a process-by-process basis depending on the goodness of fit for the power regression.

Facility-specific factors beyond ore grade will affect the result. For example, an increase or decrease in the electricity emission factor will scale the emissions intensity for that process proportionally. If the electricity emission factor is reduced by 50%, the corresponding emission intensity from electricity will decrease by 50%.

Similarly, emissions from land and air transportation can be scaled by relative transport distance; and emissions from land use may be scaled by relative disturbance area and/or the pre-existing net primary productivity in the region.

Again, this methodology is provided to assist with a first approximation of GHG emissions for facilities not included in this study. Adaptation of the results is complicated by myriad site-specific factors including but not limited to mine type, mining and milling techniques used, production rate, lifetime production, ore depth, climate, location, geology, and hydrogeology.

4.8 Scenario Analysis: Future U_3O_8 Production in SK

In the coming years, the uranium mining-milling industry in SK will be changing. Most notably, McClean Lake mill will be processing high-grade ore from the Cigar Lake underground mine. By 2018, the mine is expected to be operating at full capacity, trucking over 8,000,000 kg U_3O_8 to McClean Lake annually, at an average grade of 15.8% U_3O_8 (Cameco Corporation, 2014e). This will mark a substantial contribution to the overall production of uranium, equivalent to 73% of SK's 2013 production and 11% of global U_3O_8 production in the same year (World Nuclear Association, 2014j).

The ore grade-emissions relationship developed in Mudd (2014) puts emissions from these facilities at 30 kg CO_2e/kg U_3O_8 based on expected ore grades. By adjusting the mine-mill models used in the present study to represent the expected conditions and ore grades at Cigar Lake-McClean Lake in the future scenario, a similar estimate is obtained, albeit with a large associated uncertainty (27-74 kg CO_2e/kg U_3O_8) mainly due to the lack of precedent in milling ore with such high uranium content.

Given the large production volumes, operations at Cigar Lake-McClean Lake will have a strong influence on average emissions intensity for SK uranium in the near future.

5. Conclusion

5.1 Summary and Conclusions

This thesis presents a detailed study of life cycle greenhouse gas emissions during the uranium mining-milling phase of the nuclear fuel cycle for three paired uranium mine-mill operations in northern Saskatchewan. The life cycle analysis was conducted based on the ISO 14040:2006 standard using the Process Chain Analysis methodology. It includes a cumulative 32 years of production data from the mine-mill pairs under consideration.

The methods and results of this study address many of the recommendations and limitations indicated in a seminal meta-analysis study of nuclear life cycle analyses by Warner and Heath (2012). It achieved this by including the uranium exploration and mine-mill decommissioning phases, reporting ore grades, following ISO 14040 guidelines, identifying the primary energy mix at each facility, and using data from other existing facilities to fill data gaps.

Additionally, the study addressed a known problem in Process Chain Analysis: inclusion of "systematic errors due to the unavoidable truncation of the system boundary" (Weisser, 2007). This was done by including a very large and inclusive system boundary and low cut-off criteria of 0.1%. Furthermore, the use of the SimaPro modeling software and the ecoinvent v3.0 life cycle database allowed the inclusion of many processes failing this cut-off criteria. As a result, the systematic errors due to the use of Process Chain Analysis are minimized.

Over the study period (2006-2013 for two mine-mills, and 1995-2010 for the third), the life cycle GHG emissions of the three uranium mine-mill pairs are 84, 66, and 35 kg CO_2e/kg U_3O_8 at average ore grades of 0.71%, 1.54%, and 11.5% U_3O_8 respectively. The production-weighted average emissions are 45 kg CO_2e/kg U_3O_8 at an average ore grade of 9.12% U_3O_8. This amounts to approximately 1.2 g CO_2e/kWh electricity when considered as a phase in the nuclear power fuel cycle. Assuming that the median estimate for nuclear life cycle emissions (12 g CO_2e/kWh) in Warner and Heath (2012) is representative, mining-milling of uranium from Saskatchewan represents approximately 10% of the light water reactor nuclear fuel cycle GHG emissions.

Emissions from the full nuclear fuel cycle (12 g CO_2e/kWh), in turn, are almost negligibly small compared to emission from fossil fuel technologies like coal and natural gas with emission of 1001 g CO_2e/kWh and 477 g CO_2e/kWh respectively (OpenEI, 2012).

Due to the very high ore grades found in SK uranium deposits, it was hypothesized that the life cycle GHG emissions intensity for uranium from SK would be lower than values reported for other facilities around the world. Results for SK are lower than most, but not all, of emission intensity estimates for uranium mining-milling outside

of SK, as reported in the literature reviewed. A detailed comparison of the present study to the other, lower estimates was not conducted, as the source material was not available in English. Indeed, the detailed analysis of other uranium mining-milling life cycle studies has been difficult due to the lack of detail provided in many of the sources.

The results of this study are higher than two of the three studies included in the literature review based on SK-sourced uranium. But, because the present study uses real production data, as opposed to estimates and extrapolations, and accounts for emissions from processes often neglected in other studies, the results are more accurate.

The inclusion of more life cycle emission sources (e.g., construction, reagent consumption, employee transport) increased the overall GHG emission estimate by approximately 30% compared to the common approach of reporting emissions from energy consumption only. While this is a material contribution to the emissions total during the mining-milling phase, this finding may reduce the concern that missing emissions sources, such as mine decommissioning, materially affect emission estimates for the nuclear fuel cycle overall.

The majority of life cycle GHG emissions arise during facility operation, predominantly from electricity consumption, followed by consumption of propane and diesel fuel; transport of materials and personnel; and from emissions embodied in reagents. Remaining processes each account for less than 5% of the total and include direct and indirect emissions from exploration, construction, and decommissioning activities; reduction in existing ecosystem services due to land use change; emissions embodied in concrete, infrastructure, and equipment; direct emissions from the conversion of inorganic carbon in ore to CO_2 during the milling process; and direct emissions from wastewater and landfills.

Overall uncertainty in the result is relatively low as data was available for each of the largest contributors and their emission factors are well understood. For the production-weighted average result, the 95% confidence interval ranges from 42 to 49 kg CO_2e/kg U_3O_8. Residual uncertainty is due mostly to life cycle emission factors for electricity, carbonate content in ore, fuel economy for planes and trucks servicing the mines, and the amount of energy and materials used during construction.

The results stated above assume a provincial-average GHG emission factor for electricity generation. This factor was used because the utility does not disaggregate emission factors by region. Since northern SK is powered exclusively by hydroelectric power, the use of a regional factor would reduce estimated emissions from electricity by a factor of over 100. Using the regional factor, the GHG emissions intensity for Mine-Mills A, B, and C are 45, 48, and 20 kg CO_2e/kg U_3O_8 respectively, with a production-weighted average of 26 kg CO_2e/kg U_3O_8. This is major reduction, averaging 42%. This hydroelectric-powered GHG emissions estimate is equivalent to approximately 0.6 g CO_2e/kWh electricity when considered as a phase in the light

water reactor nuclear power fuel cycle, or 4.8% of the total for the light water reactor nuclear power fuel cycle.

The results of this study clearly show that GHG emissions from the mining-milling of SK uranium make a very small contribution to emissions from nuclear power generation overall. These results can be used to further refine future nuclear fuel cycle GHG emission estimates that include ore sourced from SK (15.7% of global supply in 2013 (World Nuclear Association, 2014j)) and may provide a framework for future life cycle studies at uranium mining-milling operations using differing processes and/or occurring in other regions globally.

5.2 Limitations

This research has a number of limitations that could be addressed with future research. The study only considers GHG emissions to air and no other emissions to soil, air, or water and so does not fully represent the environmental impact of uranium mining-milling. Furthermore, it makes no attempt to address socioeconomic factors related to the mining-milling of uranium or the nuclear fuel cycle in general.

The conditions at the facilities considered in this study may not accurately represent conditions at excluded facilities. Therefore, it may not be appropriate to extrapolate the results to estimate emissions at those excluded facilities.

This study did not include enough different mines and different types of mines to conclusively determine whether a particular mining style (e.g., open pit, underground, or in situ leaching) has systematically higher or lower emissions than another style. No in-situ mining data was included in this study.

5.3 Future Work

This study reports emissions from facilities with some of the highest ore grades currently in production. Presumably, the GHG emissions embodied in the yellowcake product will therefore be amongst the lowest in the world. In order to understand the true range of emissions from uranium mining-milling globally, it is necessary to know the emissions intensity of yellowcake produced by facilities with very low ore grades. With both numbers in hand, actual emissions from global nuclear power production will be better understood.

As a start, emissions from a number of mines with varying ore grades are collected and published in Mudd (2014). These data are not reported on a life cycle basis and may be underestimated. A life cycle study using actual production data for a low-grade uranium mine-mill is required to know whether the excluded life cycle phases make a material contribution to the emissions total.

Alternatively, this work could be expanded upon by increasing the scope of environmental impacts considered. A full life cycle analysis would include all major environmental impacts, including but not limited to: ozone depletion, global warming, acidification, eutrophication, human health, ecotoxicity, fossil fuel use, land use, and water use. This broader scope is necessary to evaluate the environmental

trade-offs of competing power sources (e.g., coal, natural gas, wind, solar and hydroelectric).

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